CONSERVATION AND MANAGEMENT OF LAKES

Symposia Biologica Hungarica

38

CONSERVATION AND MANAGEMENT OF LAKES

38

1828

Akadémiai Kiadó, Budapest

CONSERVATION
AND
MANAGEMENT OF LAKES

Edited by
J. SALÁNKI and S. HERODEK

(Symposia Biologica Hungarica 38)

Lakes are one of the most precious endowments of mankind but they have become seriously endangered by reckless human activity in the last decades. Their protection needs international cooperation. This volume contains selected papers of the Third International Conference on the Conservation and Management of Lakes, held in Keszthely, on the shore of Lake Balaton (Hungary). The book offers an overview of the global state of lake management, including the problems caused by eutrophication, acidification and toxic organic chemical pollution.

Information is given on many important lakes in Africa, Asia, Europe and America, e.g. Lakes Victoria, Baikal, Sevan and Ladoga. The case of the Laurentian Great Lakes serves as encouraging example of the international cooperation in lake conservation. Besides the ecological aspects of lake management the socio-economic, jurisdictional and institutional problems are also dealt with.

Limnologists and other ecologists, specialists and decision-makers in environmental protection and water resources management as well as students will find this book most useful.

AKADÉMIAI KIADÓ
BUDAPEST

ISBN 963 05 5583 2

Symposia
Biologica
Hungarica
38

Symposia Biologica Hungarica

Vol. 38

Redigunt

J. SALÁNKI et S. HERODEK

AKADÉMIAI KIADÓ, BUDAPEST 1989

CONSERVATION AND MANAGEMENT OF LAKES

Edited by

J. SALÁNKI and S. HERODEK

AKADÉMIAI KIADÓ, BUDAPEST 1989

Proceedings of the Third International Conference on the Conservation and Management of Lakes "Balaton '88", held in Keszthely, Hungary, 11–17 September 1988

The organization of the Conference and the publication of the book were sponsored by the United Nations Environment Programme (UNEP) and the International Lake Environment Committee (ILEC).

The designations employed and the presentation of the material in the present publication do not apply the expression of any opinion whatsoever on the part of the Secretariat of the United Nations concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of the frontiers or boundaries.

ISBN 963 05 5583 2

© J. Salánki and S. Herodek

All rights reserved. No part of this book may be reproduced by any means, or transmitted, or translated into machine language without the written permission of the publisher.

Printed in Hungary
by Akadémiai Kiadó és Nyomda Vállalat, Budapest

CONTENTS

ACIDIFICATION AND TOXIC POLLUTANTS

LAKE MANAGEMENT AS AN ECOLOGICAL, ECONOMIC AND JURISDICTIONAL COMPLEX

MODELLING IN LAKE MANAGEMENT

PREFACE

The Third International Conference on Conservation and Management of Lakes dealing with lake environment and hosted by the Hungarian Academy of Sciences and the Ministry for Environment and Water Management was held in Keszthely (Hungary) on the shore of Lake Balaton. The volume contains selected plenary lectures, and papers delivered at the Conference.

The papers represent a broad review of the state-of-the-art of lake conservation. The chapters discuss the global aspects of such important problems as eutrophication, acidification and toxic pollution of lakes, followed by case studies from four continents.

Ecologically sound lake management is a multi-dimensional problem as illustrated by the diverse range of topics discussed in this book, including the socioeconomic, institutional and jurisdictional aspects.

Our thanks are due to the International Lake Environment Committee Foundation (Otsu, Shiga Prefecture, Japan) and to the United Nations Environment Programme (Nairobi, Kenya) for their help in organizing the Conference and also for their support in publishing this volume.

The Editors

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

LAKE MANAGEMENT AND SUSTAINABLE DEVELOPMENT

G.N. GOLUBEV

United Nations Environment Programme*

The purpose of this paper is to review problems related to the management of lakes. Population growth, change of land use, economic development in the basin, impacts from outside the basin, activities on the lake itself, all these factors lead to pronounced implications of the state of lakes. The expected outputs from a lake, be it water of certain quality, expected lake level, desired fish catch, wished recreational amenities, etc. may not be provided because of the changes of the lake characteristics or of the changed and increased demand to the amenities.

Generalizing one can say that the need for lake management arises when the state and regime of the lake do not correspond to outputs expected from it. This situation is becoming quite widespread. Obviously, the objectives of lake management are always specific in each case, they are time-dependent and often very complex.

The most important consideration when addressing lake management is that a lake plays the role of an integrator of many processes going over the whole of the lake basin. Lake management includes all measures aimed at guiding and controlling the activities that affect the environmental attributes of a lake. This fact rightly turns our attention in many cases more to the basin than to the lake itself. Secondly, often international cooperation is necessary to bring about a desirable

*The views expressed herein are not necessarily those of UNEP.

change in the environmental quality of lakes, because isolated national measures, although essential, are not sufficient.

It will not be a great exaggeration to say that there is no lake in the world which does not bear the impact of man's activities. Even in the least populated and remotest areas the impacts can be detected. The following main factors are responsible for this: population growth; increasing demand for resources, including land and water; transformation of biogeochemical processes.

Over the last 30-40 years there was a steep increase in the world population. It doubled from 2.5 billion in 1950 to 5.0 billion in 1987. Most of the increase is attributed to developing countries. Population growth means increased demand for such resources as land, forest and water. The need for goods and services leads to the increase of industry, development of agriculture, expansion of transportation networks, etc. Social development, even as slow as it has been in most third world countries, exerts higher per capita demands for all the necessities of human life. All these processes are manifested in considerable changes in lake basins and the lakes themselves. It goes on everywhere, but more intensively in developing countries. Environmental management measures, including application of pollution control technologies, require access to finance and expertise which the developing countries do not have on the necessary scale. Moreover, even the state of lakes in the developing countries is not known enough.

While the population has doubled since 1950, the use of water in the world has more than tripled. By far the greatest user of water is irrigation. Since 1950 the area of irrigated land has more than tripled. That has also led to a number of pronounced problems with lakes, in particular with closed lakes characteristic of arid climates.

Exploitation of such resources as water, land or forests cannot be seen in isolation. The examples are well known. Development of irrigation exerts serious impacts on the quality and quantity of water resources. Destruction of forests provides sharp increases in soil erosion thus leading to a deterioration of land productivity and interfering with the water regime. Transformation of virgin natural landscapes into agricultural areas in the world increased soil erosion from the fields by at least five times at current levels as compared with the preagricultural period, with the corresponding noticeable increase of the river sediment transport and associated phosphorus transport (1). The value of sediment-associated phosphorus transport by rivers in the world is of the same order of magnitude as the annual global production of phosphorus fertilizers. Current eutrophication of many lakes is in fact a consequence of the above process.

No less influence on water bodies, including lakes, was and is brought about by a transformation of natural biogeochemical processes as a result of human activity. The increase of phosphorus load has just been mentioned. The same applies to other important chemical elements (nitrogen, carbon, sulphur, metals, etc.). The share of anthropogenic components of the biogeochemical cycles shows a stable upward trend. In addition, the composition of chemical fluxes in the environment is under constant change reflecting the increased inputs from human activity. Lakes, playing the role of integrating different processes going on in their watersheds, are more susceptible to chemical changes when compared with many other natural systems. As a result, we face increasing water pollution, growing salinity of water and such specific but widespread processes as eutrophication and acidification.

One emerging factor will soon play a very important role in determining the state of lakes. I am talking about the coming climate change. It is expected that before the middle of the next century the average temperature of the earth will be higher by 1.5-4.5°C than it is now. The reason is the build-up in the atmosphere of carbon dioxide and other greenhouse gases due to human activity. It will be the largest temperature increase during the existence of man.

The implications, in particular for lakes as integrators of processes over the whole watershed, are enormous. Changed natural landscapes, types of agricultural practices, water balance and the hydrologic regime would exert great influence on many different patterns of the state of lakes in all parts of the world. Elaboration of a long-term strategy for the management of lakes cannot proceed without due account being taken of the consequences of climatic change. In view of the many uncertainties and our incomplete knowledge of the problem we should develop an adaptive, flexible strategy.

Lakes are parts of larger terrestrial ecosystems and they depend on local natural conditions to the same extent as the ecosystems themselves. Therefore, a wide variety of lake regimes can be observed in the world. The combination of natural regimes with man-made impacts provides a very rich diversity of actual or potential problems which we should address in the process of lake management. Many problems with lakes have been widely publicized. The lakes are the focus of the processes present in their watersheds but they also became the focus of public attention because they reflect those human actions which disregard the environmental concerns related to the lakes and their basins. The following are a few well-known examples of the problems.

The North American Great Lakes are the largest freshwater ecosystem in the world. Their basin concentrates about one-sixth of the US national income and about one-third of the national income of Canada. The considerable population, diversified industry and intensive agriculture have led to water pollution. Two major problems are present: eutrophication and pollution by toxic chemicals. Lake Erie has faced the most severe pollution problems because its basin contains the largest population of any of the lakes and is quite heavily industrialized while having the smallest volume of water. Basically, the problems are typical for many other lakes situated in industrialized countries with temperate climates.

The Great Lakes provide an example of quite successful management, exercised jointly by the United States and Canada. Control of eutrophication is pursued through the management of phosphorus loadings, in particular by reducing phosphorus in municipal effluents and by limiting the phosphorus content in household laundry detergents (2). The phosphorus loads demonstrated a considerable decrease in the seventies followed by a less pronounced decrease in the beginning of the eighties. Municipal phosphorus loads of Lake Erie in the decade 1972-1982 decreased more than five times.

Management of toxic substances started with the development of an information base and monitoring system. In 1986, the Toxic Substances Control Agreement was approved by eight U.S. states and two Canadian provinces. The agreement delineates a number of management actions in order to reduce the toxic loadings (3). Considerable improvement in the quality of the state of the Great Lakes, especially of Erie and Ontario, is due to a comprehensive, multidisciplinary and multiobjective approach particularly during the last 10-15 years.

Lake Balaton in Hungary demonstrates very clearly a eutrophication control problem. For the last 30-40 years this large (596 km^2) and shallow (average depth 3.1 m) lake has been receiving an ever increasing amount of phosphorus (4). Since 1950 the fertilizer application in the basin has been increased 60-70 times. Very large animal farms did not exist in 1950; now they contain about 100,000 animals. Between 1950 and 1978 tourism increased 14-fold. Development of sewerage and waste water treatment did not keep up with the rate of economic development. It is estimated that the phosphorus load of the water body has increased by an order of magnitude during the past 20-25 years (4). This circumstance accelerated eutrophication of Lake Balaton.

A comprehensive programme of lake management is under way. Its main objective is the sharp reduction of phosphorus loading coming from the watershed. However, shallowness of the lake provides inputs of phosphorus from the bottom sediments. Even if the external load is removed, the internal load from the sediments would, in the short run, be about half of the original level. The problem will not be discussed here any longer, since a number of presentations on the problem may come forth during the Congress.

The problem of acidification of lakes has received much publicity. It is known in forested and mountainous regions with temperate climate, particularly in Europe. In Sweden, out of

85,000 lakes acidification affects 15,000. Some 4,500 lakes have almost no fish life and 1,800 lakes are so badly acidified that they are close to lifeless (5).

The main reason for the problem is the deposition of sulphuric acid components. The solution of the acidification problem goes well beyond the lake basin and even beyond national boundaries creating the need for international management approaches. Local management of acidification is costly; each year in Sweden several thousand lakes receive more than 100,000 tonnes of lime at a cost of nearly 100 million kronor (5). However, this is not a permanent solution since most of the bicarbonate ion delivered to the lake is used up in two or three years.

Not going into details of the lake acidification problem, since there is a special presentation on it, it may be noted that the case of the Swedish lakes is a good example of the need for an international approach to the management of small national lakes.

Lakes in tropical countries pose a problem which is much less pronounced, if it exists at all, in temperate climate areas. The problem concerns water-borne diseases like schistosomiasis, malaria, onchocerciasis, etc. These diseases are associated with the vectors living in water or near water. Construction of man-made lakes usually leads to an increasing spread of such diseases as schistosomiasis and malaria. However, in the case of water reservoirs there are indications that no dramatic increase in schistosomiasis occurs when the annual drawdown of a man-made lake is large (6).

Lake siltation is another widespread problem. The rate of sediment transport causing siltation has increased considerably mainly due to deforestation and agricultural practices not paying regard to soil conservation. Perhaps, the threat of siltation is more typical of water reservoirs in developing countries. The Dal lake in Kashmir, India, has shrunk from 21.0 km^2 to 12.4 km^2 over the last 40-50 years, the main reason being heavy sedimentation amounting to 70,000-80,000 tonnes a year (7). The reservoir of the Ambuklao Dam in Luzon in the Philippines has been affected so much that its useful life has been reduced from 60 to 32 years. The reason is uncontrolled deforestation in the basin (8). Management of the siltation problems depends upon effective land management in the basin.

Areas without drainage to the ocean or so-called closed areas, occupy 32 million sq.km or 21% of the land surface. Normally, lakes here are the final collectors of water from their basin. They are particularly sensitive to the oscillations of the water balance components, and, hence, to the economic developments in the basin.

The Aral Sea is the extreme and the most dramatic case. Before 1960, the water level of the sea mostly depended upon natural factors. It fluctuated within one meter having a volume of over 1000 km^3 and an area of about 66,000 km^2. The only two tributaries, Amudarya and Syrdarya, had their total flow at the border between the mountains and the plain of about 110 km^3 a year. Because of irrigation, transpiration of phreatophytes and other less important factors, the Aral Sea received then from the two rivers 56 km^3 on the average. Massive development of irrigation in the basin considerably increased consumption of water resources. In many cases the canals were not lined and huge amounts of water had infiltrated. In addition, the application of water to the fields was excessive. These main factors led to the situation when Syrdarya ceased to have a constant flow at its mouth and Amudarya discharged a small fraction of its natural flow. By 1978, the flow was down to 31 km^3. Now, ten years later it is about 11 km^3 (9). As a result, the water level dropped by 12 m by 1985 and continues to go down. Correspondingly, its area and volume have shrunk considerably. Salinity rose from 10.1 to 22 g/l and continues to increase. The unique lake ecosystem has been destroyed and the once prosperous fisheries do not exist any more. The shoreline receded by dozens of kilometers and newly exposed salty land serves as a source of salts deflated by wind to the neighbouring territories. The whole phenomenon is now called an "ecological disaster" (10).

The main mistake or shortsightedness was that the basin and the lake were not considered as a unified system. In cases like this there is no perfect solution to the problem. However, the optimal parameters of the Aral Sea and the water consumption drawn from projections of economic development in the basin could have been found if a multicriterial analysis had been applied and a management strategy had been developed. The overall objective for irrigation development was the growth of agricultural production for the country coupled with social and economic development of the region. Disregard of major environmental considerations slowed down achievement of the main objective.

The brief review of the few cases made above exemplifies the main problems to be addressed through lake management (chemical pollution, eutrophication, acidification, water-borne diseases, siltation, drop of water level and increase of salinity). The list of the problems mentioned here is not exhaustive. Apparently, management of the lakes themselves may not bring solutions to the problem and in any case would be feasible for small and very small lakes only. In most cases, and for most lakes, their management has to be carried out in the framework of the lakes' basins.

Water reservoirs are a special category of lakes where obviously there are more possibilities for in-lake management. Their share in the total number and volume of lakes in the world is going up. The total volume of water reservoirs in the world exceeds 6,000 km^3 and the area is 380,000 km^2 (11). It would be misleading to compare these figures with those for lakes because the very large lakes would dominate the picture. The largest reservoirs by volume are less than 200 km^3 (Bratsk, 169 km^3; Kariba 160 km^3; Nasser 157 km^3; Volta 148 km^3). If we deduct from the total volume of freshwater lakes (91,000 km^3) the volume of the 16 largest lakes exceeding 200 km^3 each (12) the difference would be 11,500 km^3. Thus, the volume of water in man-made lakes is about 50% that of the world lakes, excluding the largest ones. This figure shows the growing ability of man to actively manage water bodies.

The above discussion leads us to a not very new conclusion: a multicriterial and multidisciplinary approach to lake management is the only tool to achieve management objectives on a long-term basis. The approach is also often called integrated.

It can also be named multidimensional. The approach stresses the need to combine technical, economic, legal, institutional and political measures because none of the measures taken in isolation usually can provide an effective and lasting solution. There are many publications introducing or supporting this approach. On appearance, everybody is agreeing with it and yet it does not always work. Perhaps, the large number of publications calling for integrated lake management may also serve to illustrate the fact that the approach is not yet practised widely.

The question why it is so, may be one of the main questions for this Congress. In the next part of this paper I am going to discuss a few reasons why the integrated approach does not always work.

The first group of reasons could be named insufficient knowledge and uncertainty. To get a clear picture of the state of a lake in question means knowing the main processes and factors such as water balance, load and loading tolerance and mass balance for key chemical components, state of biota in the lake ecosystem, etc. It requires skills, equipment and/or funds which may be lacking even in an advanced country, let alone developing ones. Such data as those on historical, current and future land use, agricultural practices, sewage and industrial waste water treatment are not very easy to obtain either. Even less is known about impacts of human activities in the basin on the state of a lake.

And yet, problems with lakes increasingly press for solutions. Under these circumstances there is a requirement to operate on the basis of insufficient and poor knowledge which creates uncertainties. Other reasons behind uncertainty are: stochastic character of many natural processes, in particular those of hydrological and meteorological nature; likely climate change and its implications; and short-term horizon for societal goals. An additional, but very important dimension of the uncertainty factor is insufficient awareness of the decision-makers, industry representatives and public at large of the interdependence between human actions and their impacts on the state of a lake.

Traditionally, technical (engineering) methods of management have been the main tool in solving water-related problems. However, there are so many examples of failure when a purely engineering approach has been applied. It can be said that the technological means of lake management are at the centre of action, but if applied without the right kind of economic, legal and institutional support they bring more problem than solutions.

Economic factors for the management of lakes are not yet fully used. For water, although a scarce resource in many, if not in most of the countries, there are no charges for its use at all, or they are well below costs. Penalties, taxes, indemnities and other "punitive" economic mechanisms oriented to regulate against water pollution are not used everywhere, particularly in developing countries. Corrective measures on lake basins are difficult to relate in economic terms to the improvements in the state of a lake. Much better knowledge is needed to know how economic tools can help in the process of integrated lake management.

Very important, if not the most important are the institutional components in multidimensional lake management. Different interest groups having their own particular objectives are often represented by different pieces of governmental machinery. Often, those objectives are in conflict. For example, the optimal level of a lake may be seen in a different way by a fishery department and an irrigation department. Multipurpose water reservoirs in particular are an area of conflict among institutional groups representing different sectors of the economy. Another source of institution-related conflicts is the difference of local (regional) and national objectives of lake management.

A very serious source of problems in conducting multiobjective lake management is an imperfect governmental structure and its modes of operation. Environment-related laws and regulations are not always in existence and even if they exist they are not always complied with. Environmental ministries/agencies in many countries do not yet play an effective role in integrating environmental issues in economic planning and management. Allocation of resources is made without proper regard to the interests of long-term sustainable development. The involvement of ministries of finance and planning and/or of higher levels of coordination and direction into planning and execution of integrated lake management is therefore very important.

No less important are the political factors of integrated lake management. The environment has become a very prominent consideration in the political life of many countries. The public is, and should be, involved in the process of decision-making concerning integrated lake management and in the monitoring of its implementation. Non-governmental organizations are notably instrumental in this process.

The management of international lakes provides a special and very complicated dimension of the whole issue. Unlimited use, or abuse, of these shared waters is no longer possible without adversely affecting the use of the same water body by neighbouring countries. In many places there is an urgent need for bilateral, multilateral, and global agreements concerning the management and development of internationally shared waters.

Numerous bilateral and multilateral treaties have been adopted between states; and yet, the existing rules of customary international law prohibiting pollution or consumption which causes substantial damage to the interests of other riparian states on an internationally shared water body are often not implemented in practice. While individual nations are giving teeth to their legislation concerning pollution from chemicals, sewage, thermal discharge, radioactivity, and salinity, international co-operation is lagging behind. The number of national or international adjudications of disputes concerning pollution or other aspects of the use of international water bodies are few. Not surprisingly, the rules of customary international law pertaining to the management of international water bodies are difficult to put into practice.

At the end of the last century, the theory of "absolute territorial sovereignty" was supported by some courts. Accordingly, each state had the sovereign power to do what it liked with or on its territory, regardless of the result. More recently, "absolute territorial sovereignty" has been tempered by the contrary view that a state may do nothing within its territory which may produce harmful effects within the territory of another state. Most legal writers have now adopted a compromise of these two extremes concerning inland waters: a state should act in such a way as to avoid causing appreciable and unreasonable harm to the territory of a neighbouring state. Essentially the international "Good Neighbour" principle, this compromise is also found in Principle 21 of the 1972 Stockholm Declaration on the Human Environment (13).

The Stockholm Conference discussed several aspects of the management of shared natural resources applicable to shared waters. Principles 21, 22 and 24 of the Declaration are predicated on the tripartite concept of sovereignty, responsibility and co-operation. Those principles are also contained in the 1982 U.N. General Assembly World Charter for Nature and are a part of several international agreements and conventions regarding the use of international water bodies, including the 1978 agreement between Canada and the United States on the Great Lakes Water Quality.

Comprehensive and successful protection of shared waters required that the "Good Neighbour" principle be fully implemented. Established principles of "no substantial harm" and "equitable and reasonable utilization" are also adaptable to international norms which should be applied to shared water bodies. Finally, the principles of sovereignty, responsibility and co-operation need to be established as international legal norms. However, the successful management of internationally shared lakes involves more than just the establishment of laws, their enforcement, and compliance. Management and sustainable development require: co-operation and common planning; environmental impact assessments and emergency action plans; prior consultations and notifications before national action; planned common development and information exchange; and mechanisms for successful dispute settlement. It is a challenge for international organizations like UNEP to help governments to develop common multiobjective management action plans.

Clearly, integrated or multiobjective approach to lake management is based on interdependence of the dynamic actions. The philosophy and general framework of sustainable water development and management are addressed by A.K. Biswas and J. Kindler in a special publication prepared on behalf of UNEP (14). Guidelines on principles and concepts of lake management are in a final stage of preparation by S. Jorgensson (15) under the auspices of the International Lakes Environment Committee with the support of UNEP. It is hoped that these publications will make a noticeable contribution to promoting the multiobjective lake management approach.

The integrated lake management approach will be successful only if it is a part of territorial (regional, national) planning and management. Strictly speaking, it should be a component of basin-wide planning and management. This in turn should be backstopped by a strategy of sustainable development "... to ensure that the needs of the present are met without compromising the ability of future generations to meet their own needs" (16). As stated in the Brundtland's Commission Report, "...sustainable development is not a fixed state of harmony, but rather a process of change in which the exploitation of resources, the direction of investments, the orientation of technological development, and institutional change are made consistent with future as well as present needs". The issues of integrated (multiobjective) lake management discussed above fit precisely into this explanation.

There is inherent conflict between economic development in the basin of a lake and the desired state of the lake. The strategy of sustainable development should help to attain a feasible solution of the conflict. From the economic theory of natural resources exploitation it is known that the cost of protecting the environment needs to be balanced against the perceived benefits (17). That means once again that one needs to establish an expected state of a lake as a compromise among many choices keeping in mind a balance between social costs and benefits. Not all costs and benefits are quantifiable, however.

Such choices are very hard to make. A typical problem is the elaboration of the right water management policy for the basin. "Hydraulic-engineering type of thinking" prevails in many cases. It is not manifested in a straightforward way. Almost nobody now dares to speak against the need to consider environmental issues in water resources planning or in designing a new project. However, very often environment-related aspects are conceived as an addition, or an afterthought, and not as an integrated part of the whole scheme. And when it is an addition it is liable to receive less funds, less attention, up to complete contempt. Interweaving environmental considerations into technological, legal, economic and institutional components of regional (basin-wide) management is the only effective strategy.

In order to define multiobjective lake management strategy based on the concept of environmentally sound sustainable development one needs to consider also social, economic, cultural, and aesthetic values. Noninferior options of the strategy then can be found. It is such an option "... where any improvement in one objective can be achieved only at the expense of degrading another" (14). "Selection of the best compromise solution from a set of noninferior solutions requires that trade-offs between achievement of different objectives are identified and evaluated.... This is undoubtedly the most difficult stage of multiobjective analysis" (14).

In spite of all the difficulties some success stories exist. Perhaps, the best example known to me is the management of Lake Biwa, Japan. The state of the lake is maintained within established parameters due to a very elaborated and costly multiobjective management scheme. It seems that a compromise between economic development of the basin and the state of the lake has been achieved (15, 18). This balance is dynamic and requires constant management and careful planning.

It is expected that other success stories will be brought to the attention of this Congress. Each case, successful or not, is unique. But the experience, accumulated and digested, is our common heritage.

REFERENCES

1. Golubev, G.N. Soil Erosion and Agriculture in the World: An Assessment and Hydrological Implications. IAHS Publication No. 137, 1983.
2. Environmental Quality. The 15th Annual Report of the Council of Environmental Quality, USA, 1984.
3. World Resources, 1987. A report by IIED and WRI, USA.
4. Somlyody, L. and van Straten, G. Modelling and Managing Shallow Lake Eutrofication. Springer-Verlag, 1986.
5. Acid Magazine, No. 1, 1987, Sweden.
6. Stanley N.F. and Alpers M.P. (eds) Man-made Lakes and Human Health. Academic Press, London, 1975. (Cit. from: P. Bolton. Mozambique's Cabora Bassa Project: An Environmental Assessment. In: E. Goldsmith and N. Hildyard (eds) The Social and Environmental Effects of Large Dams, 1986.)
7. Kaul, V. Indian Lake Environment and Conservation, In: Shiga Conference 1984 on Conservation and Management of World Lake Environment. Otsu, 1985.
8. Tolba, M.K. Caring for Lakes in a Developing World: Minimum Environment with Sustainable Development. Ibid.
9. Babich, B.I. and Grigoriew, E.G. Protection and Rational Use of Water Resources (in Russian). Publ. House "Znanie", Moscow 1987.
10. Morgun, F.T. Statement at the XIXth Conference of CPSU (in Russian). "Pravda", 2 July 1988.
11. Voropaev, G.V. and Avakian, A.B. (eds) Reservoirs and their Environmental Impacts (in Russian). Moscow 1986.
12. World Water Balance and Water Resources of the Earth. USSR Committee for the International Hydrological Decade, 1974. English Translation: UNESCO 1978.
13. Declaration of the United Nations Conference on the Human Environment. Stockholm, 1972. In: Report of the UN Conference on the Human Environment. United Nations, A/CONF.48/14/Rev. 1, New York 1973.
14. Biswas, A.K. and Kindler, J., Sustainable Water Development and Management. UNEP, 1988 (in print).

15. Jorgensson, S. Guidelines on Environmental Management of Lakes. ILEC/UNEP, 1988 (in print).
16. Our Common Future. World Commission on Environment and Development. Oxford University Press, 1987.
17. Holdgate, M.W. A Perspective of Environmental Pollution. Cambridge University Press, 1979.
18. Sueishi, T. The Current Status and Problem to Lake Environment Administration, Focussing on the Case of Lake Biwa. In: Shiga Conference 1984 on Conservation and Management of World Lake Environment, Otsu 1985.

EUTROPHICATION AND ITS CONTROL

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

GLOBAL PROBLEMS OF EUTROPHICATION AND ITS CONTROL

R.A. VOLLENWEIDER

NWRI-CCIW Burlington, Ontario, Canada

EUTROPHICATION

1. INTRODUCTION

Accelerated eutrophication of waters, both fresh and marine, as experienced during this century in most parts of the world, represents a serious degradation of water quality. This, in turn, interferes with most economic losses. Also, impairment of water quality due to eutrophication can lead to health-related problems though the connection between the two complexes may at times be obscured by indirect vectorial relationships.

Eutrophication and oligotrophication of waters occur in part as natural slow processes over centuries, yet such long-term variations do not necessarily cause concern. Man-depending accelerated eutrophication, on the other hand, has been developing rapidly over the last few decades an all highly populated and industrialized countries and has become an increasing threat to lakes, water reservoirs, marine coastal areas. This evolution has triggered several cooperative studies over the last two decades, nationally and internationally, and concerted efforts to correct the problem. However, though, locally, eutrophication of specific water bodies is now under control, these efforts have only been partially successful. Eutrophication, at the global scale, is still one of the most serious water quality problems, and continues to increase in many parts of the world.

2. PROBLEMS AND IMPAIRMENTS CAUSED BY AND/OR ASSOCIATED WITH EUTROPHICATION

The most common practical problems caused by eutrophication of lakes, reservoirs and impoundments are listed in Table 1 under four headings: (a) water quality impairment for drinking water purposes; (b) recreational impairments; (c) impairments of fisheries; and (d) impairments of holding capacity. Some of these problems (though not all) can also be linked to other causes such as mineral turbidity, humic substances, low dissolved solids and thermal discharges. However, in comparison to the effects of eutrophication, these occurrences are less frequent.

Health problems to bathers are due to high pH (dermatitis, conjuntivitis), swimmer's itch (schistosomiasis), in warm climate also bilharziasis, and more rarely diarrhea due to ingestion of toxin producing

Table 1. Problems associated with lakes, reservoirs and impoundments

Problem areas	Caused by, or indirectly dependent on				
	Algal blooms and species composition	Excessive macrophyte and littoral algal growth	Altered thermal conditions	Mineral turbidity	Low dissolved solids
Water quality impairment:					
- taste and odour, colour, filtration, flocculation, sedimentation and other treatment difficulties	very frequent	at times	at times	at times	-
- hypolimnetic oxygen depletion, iron, manganese, CO_2, NH_4^+, CH_4, H_2S, etc. formation	frequent	at times	at times	-	at times
- corrosion problems in pipes and other man-made structures	frequent	at times	-	at times	at times
Recreational impairment:					
- unsightliness	frequent	at times	-	at times	-
- hazard to bathers	-	frequent	-	-	-
- increased health hazards	at times	at times	at times	-	-
Fisheries impairment:					
- fish mortality					
- undesirable fish stocks	frequent	frequent	at times	at times	-
Aging and reduced holding capacity and flow:					
- by silting	at times	at times	-	frequent	-
- pipe and screen clogging	-	at times	-	-	-

algae. Schistosomiasis and bilharziasis are indirectly connected to macrophytes which offer substrate and nourishment for cercaria carrying snails and water fowl.

In marine waters, the occurrence of toxic algae (certain dinoflagellates) can be a serious health hazard when infected bivalves are consumed (paralytic shellfish poisoning). Also, massive occurrence of zoeae (larval forms of certain crustaceans) have been linked with dermatitis cases.

Toxic fresh water blue-green algae are less a threat to man than to cattle (livestock poisoning). However, excessive buildup of nitrates in drinking water is the cause of infant methemoglobinaemia.

In tropical waters, and often in rivers, macrophyte growth can be a serious obstacle to navigation and fisheries. The occurrence and massive proliferation of *Eichhornia crassipes*, flowing mats of *Lemna* and *Salvinia* are among the most serious nuisance in warm waters. From its original geographic region in tropical South and Central America, Eichhornia has invaded during this century the warm water belts of all continents. While not necessarily due to eutrophication alone, this species is now to be found in all kind of waters, from soft fresh waters to slightly brackish waters.

A cautionary note is here in place in regard to eutrophication. High productive waters are not necessarily the effect of man-induced eutrophication, but can be natural. Such naturally high productive waters (both fresh and marine) sustain often also a high fish production. What is in question here is the process of eutrophication above and beyond the natural state of waters which then leads to undesirable conditions. This, however, does not exclude the possibility that conditions and quality characteristics of naturally high productive waters may be felt as undesirable for specific water uses.

3. DEFINITION

Eutrophication is defined as the enrichment of waters with plant nutrients, primarily phosphorus and nitrogen (cf. below), leading to enhanced plant growth (both algae and macrophytes) which results in visible algal blooms, floating algal and/or macrophyte mats, and benthic algal and submerged macrophyte agglomerations. When decaying, this plant material leads to the depletion of the oxygen reserves of bodies of water which, in turn, causes an array of secondary problems such as fish mortality, liberation of corrosive gases and other undesirable substances, such as CO_2, CH_4, H_2S, organoleptic substances (taste and odour), toxins, etc.

Although the more specific reaction of bodies of waters depend on climatic, limnological, oceanological and other conditions, the symptoms and manifestations of eutrophication as described are similar in all kinds of waters, fresh and marine. In shallow waters, macrophyte growth may (but not necessarily) be predominant over phytoplankton. In warm climate regions (tropics and subtropics), the manifestations may be more severe.

4. ASSESSMENT CRITERIA

Assessment of trophic conditions and classification of bodies of water can be made using either qualitative criteria only, or using also quantitative characteristics. Both approaches pose certain problems not easy to overcome. Among the problems to be mentioned are: None of the criteria, used singly or in combination, can satisfactorily be applied to all classes of waters. Bodies of waters of warm climate regions are not necessarily comparable to temperate and cold region waters. Also, brackish and marine waters may require different criteria. For lakes and reservoirs of temperate climates, the criteria are relatively settled and accepted, yet, there still remain uncertainties to how many parameters need be considered. Criteria appropriate for running waters are still lacking. Difficulties also exist in satisfactorily evaluating bodies of waters carrying high mineral turbidity, or which are polluted by factors others than those related to eutrophication.

4.1 Qualitative characteristics and criteria

For lakes and reservoirs, the qualitative criteria are relatively straightforward and applicable over a wide range of cases. The most important are summarized in Table 2, referring to five classes. However, in the present report, only three class denominations are used by combining classes 1 and 2 (ultraoligotroph-oligotroph) and classes 4 and 5 (eutrophic-hypertrophic). The rational for this is practical: classes 1 and 2 lakes normally pose few problems, while the cases belonging to classes 4 and 5 are those of major concern. Lakes of the transitional class 3, may or may not pose problems. The diagnostic value of class 3 lies in the fact that signs of mesotrophy are often indicative of a more serious process of increasing eutrophication.

To a limited degree, the criteria of Table 2 can also be used for flowing and marine waters. However, in rivers, the oxygen regime characteristics may be at variance with those pertaining to lakes and reservoirs in the sense that oxygen variability over a 24-hour cycle is often more indicative than its vertical variability. In coastal marine waters, on the other hand, the extent to which the oxygen regime relates to eutrophication depends largely on the extent and variability of salinity stratification. Under conditions of strong density stratification along relatively shallow coastal areas and embayments, pronounced oxygen depletion can occur causing extended fish mortalities.

4.2 Quantitative characteristics and criteria

For lakes and reservoirs, the OECD study has come forward with two ways of approach: a fixed boundary and an open boundary system. Numerical values derived for the fixed boundary system are given in Table 3, and are to be used as guides, i.e. as indicative. This means that transitions between different characteristics are possible. The open boundary system which accounts better for such uncertainties, is not based on fixed figures, but rather on coincidences of probabilities of characteristics of a certain class. This approach has now also been used for coastal marine areas.

The OECD system applies to temperate region lakes and reservoirs; outside such climates the figures and criteria may have to be modified, e.g. in warm climate regions chlorophyll values may likely have to be increased. Also, even under oligotrophic conditions, hypolimnetic oxygen may be low.

The shortcoming of the OECD system, however, is not only in this, but also in the fact that other limnological characteristics are not sufficiently accounted for. A slightly different system has been developed for the Baltic Belt lakes (cf. Hillbricht-Ilkowska, 1984). The merit of

Table 2. Trophic characterization of lakes and reservoirs

Limnological characterization categories	Ultra-oligotrophic	Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Biomass	very low	low	medium	high	very high
Green and/or blue-green algae fraction	low	low	variable	high	very high
Macrophytes	low or absent	low	variable	high or low	low
Production dynamics	very low	low	medium	high	high, unstable
Oxygen dynamics:					
epilimnic	normally saturated	normally saturated	variably saturated	often over-saturated	very unstable varying from high-over
hypolimnic	normally saturated	normally saturated	variably under-saturated	under-saturated to complete depletion	saturation to complete lack
Impairment of multipurpose uses	low	low	variable	high	very high

Table 3. Boundary value criteria for trophic categories according to OECD 1982 (fixed boundary system)

Trophic category	P	ch	max ch	sec	min sec	Oxygen % saturation over ground depending on mean depth
	(mg/m^3)			(m)		
Ultra-oligotrophic	4.0	1.0	2.5	12.0	6.0	90%
Oligotrophic	10.0	2.5	8.0	6.0	3.0	80%
Mesotrophic	10 - 35	2.5 - 8	8 - 25	6 - 3	3 - 1.5	40% to 89%
Eutrophic	35 - 100	8 - 25	25 - 75	3 - 1.5	1.5 - 0.7	40% to 0%
Hypertrophic	100	25	75	1.5	0.7	10% to 0%

P^Y = mean total phosphorus

ch^Y = mean yearly chlorophyll

max ch = chlorophyll maxima

sec = mean yearly Secchi disc transparency

min sec = Secchi disc transparency minima

*There can also be pronounced mesolimnetic oxygen maxima or minima depending on thermal stratification.

14ese approaches lies in the fact that, depending on a variety of limnological factors, it permits not only a trophic characterization, but also an indication of susceptibility to eutrophication of different lake types. However, the applicability to lakes outside the Baltic must still be tested.

5. ANALYSIS OF THE CAUSAL RELATIONSHIPS

5.1 General consideration

As stated in #3, nitrogen and phosphorus availability in both fresh and marine waters is the factor complex mostly responsible for determining the level of productivity of waters in terms of phytoplankton and macrophyte biomass. However, biomass is not solely composed of nitrogen and phosphorus. Therefore other factors must also be considered and their relative importance established. Classically, the elemental composition of biomass contains the following constituents, listed in four groups:

(1) C, N, H, O, P, S;
(2) Ca, Mg, K;
(3) Fe, Mn, Co, Mo, Si; and
(4) Na, Cl, Al, Cu, Zu, V.

The elements listed in groups 1 to 3 are ubiquitously required while the elements in group 4 may or may not be required. These elements therefore can immediately be discarded as factors involved in eutrophication. According to all knowledge, the elements in groups 2 and 3 are sufficiently available in most waters, although some limiting conditions may at times exist. This applies also to some organic growth factors, such as vitamins, mainly of the B complex (cobalamin, thiamine and biotin) required by many planktonic algae and macrophytes. This reduces the discussion essentially to group 1. Hydrogen and oxygen can be excluded, and sulphur is not known to be limiting. However, carbon can temporarily be limiting in very soft freshwaters, or at high pH. Hence, the elements of importance remain nitrogen and phosphorus.

Although there is some variability between different types of biomass, using the elements of group 1, one can formulate some basic reaction equations of formation and mineralization. For phytoplankton, one possible equation which accounts for the composition analytically found would be

$$126\ CO_2 + 16\ NH_4^{2-} + SO_4^{2-} + 78\ H_2O$$
$$C_{126}N_{16}H_{208}O_{60}(PO_4)(SH) + 12\ H^+ + 137\ O_2$$

or, if instead of NH_4^+, NO_3^- is used as nitrogen source

$$126\ CO_2 + 16\ NO_3^- + HPO_4^{2-} + SO_4^{2-} + 114\ H_2O$$
$$C_{126}N_{16}H_{208}P_{60}(PO_4)(SH) + 20\ OH^- + 169\ O_2$$

From this, one can infer that 31 mg P/m^3 and 224 mg N/m^3 are required to produce a phytoplankton (ash-free) dry mass of approximately 3000 mg/m^3, corresponding to a phytoplankton fresh mass of roughly 15 000 mg/m^3 or 15 cm^3 plankton volume/m^3 with a chlorophyll content of between 20 to 30 mg. These are potential peak values which can be expected in waters containing some 30 mg $P(PO_4)$ and 200 to 300 mg $N(NO_3, NH_4)/m^3$.

Further, one can deduce from the above that if the N/P ratio in waters is essentially larger than 7 to 10, phosphorus becomes the limiting factor, and conversely, if the N/P ratio remains essentially below 7, nitrogen becomes the limiting factor.

The relationship in macrophytes and some benthic algae, however, may deviate from the above, mostly because of a higher carbon content per unit of fresh weight. While in phytoplankton biomass the C/N ratio varies between 5 to 8, in sessile algae it can be as high as 20, and in macrophytes as high as 50 to 100. It is therefore more difficult to predict the potential biomass of sessile algae and macrophytes. For Eichhornia, which is nitrogen- rather than phosphorus-controlled, the nitrogen content per fresh weight ranges between .05 to .25% (1 to 5% N/dry weight), and the N/P ratio from 3 to 6. One g of nitrogen, therefore, can produce between 400 to 2000 g fresh weight, i.e. 6 to 30 times more than would result as phytoplankton wet weight.

The reaction equations given above, read in the reverse sense, provide also information on the amount of oxygen used for complete mineralization. 3000 mg phytoplankton dry weight (or 1500 mg or 15 cm^3 fresh biomass) would require between 4400 to 5400 mg O_2, corresponding to 1.5 to 1.8 g O_2/g dry weight. This explains the dramatic effect which eutrophication can have on the oxygen regime of waters.

5.2 Application to lakes and reservoirs

The OECD Cooperative Programme on Eutrophication, conducted on some 200 lakes and reservoirs over the temperate region of Europe, North America and Japan, including a few lakes from temperate subtropical regions, has shown that in a majority of cases phosphorus is the controlling factor. For the purpose of substantiating this claim, the data assembled by Lee and Jones (1986) have been reelaborated and are presented in form of a contingency table (cf. Table 4). The breakdown in subclasses is based on the OECD fixed boundary classification (cf. above), and the number of cases have been counted for each square. With this, a too rigid definition of the relationship which would result from a simple regression calculation is avoided and, hence, more open to interpretation. More than 50% of the cases (185 out of 335) scatter directly along the main axis which indicates a very high correlation between production level (measured as annual mean of chlorophyll) and phosphorus supply (estimated from $P_j/(1+\sqrt{\tau_w})$, i.e. the mean inflow concentration of phosphorus modified by a statistical term of the flushing rate).

Another 18% (59 out of 335) lies above this main axis, while 27% lies below. This latter may give raise to the question whether factors others than phosphorus control these lakes, or at least strongly co-determine their production level. Indeed, there are some cases among those lakes for which it was likely that nitrogen insufficiency reduced the mean annual biomass level. Noticeably, potential nitrogen limitation was reported for New Zealand lakes (White, 1983). In other cases (e.g. Lake Lugano, Italy-Switzerland; and Esrom, Denmark) the N/P ratio was generally low. Other reasons for lower than average chlorophyll levels are light limitation by either mineral turbidity, humic substances, deep mixing, or potential effects of zooplankton grazing.

Table 4. Statistical relationship between normalized yearly phosphorus load and mean annual chlorophyll concentration - 365 case studies (original data from OECD and Jones and Lee, 1986)

Chlorophyll (mg/m^3)	"P_λ" = $P_J/(1+\sqrt{\tau_w})$ (mg/m^3) < 4	4-10	10-35	35-100	> 100	Σ cases (chlorophyll)
> 25	-	-	2.3 1 1.0	25.0 11 16.2	72.7 32 48.5	44 13.1
8-25	-	2.7 2 2.3	14.7 11 11.2	44.0 33 48.5	38.7 29 43.9	75 22.4
2.5-8	-	23.3 27 30.7	51.7 60 61.2	20.7 24 35.3	4.3 5 7.6	116 34.6
1-2.5	8.4 7 46.7	62.7 52 59.1	28.9 24 24.5	-	-	83 24.1
< 1	47.1 8 53.3	41.2 7 8.0	11.8 2 2.0	-	-	17 5.1
Σ cases (phosphorus)	4.5 15	26.3 88	29.3 98	20.3 68	19.7 66	(100) 335 (100)

6. THE STATE OF EUTROPHICATION WORLD-WIDE

A precise assessment of the extent of eutrophication worldwide, covering all possible cases, is, though quite difficult, not impossible. Tables 5a and 5b provide a generic overview of the relative importance of eutrophication in fresh waters (lakes, reservoirs, rivers and canals, and coastal areas). This information has been extracted from scientific publications, published and unpublished reports. No claim can be made that all situations are adequately covered. The fact that Europe is treated in more details does not mean that eutrophication in this continent is more serious than elsewhere. It reflects simply the availability of more detailed data. Also, more information is available on lakes and reservoirs than on other bodies of water. Nonetheless, these summary tables reflect sufficiently the seriousness of the problem worldwide.

Table 5a. The eutrophication problem in the European countries. Modified from Vollenweider (1979) and Vighi & Chiaudani (1986)

	Natural lakes	Reservoirs, rivers and irrigation systems	Estuaries, lagoons	Marine coastal waters
Austria	++			
Belgium		+		
Czechoslovakia		+		
Denmark	++			++
Finland	++		+	+
France		++	+	++
Germany (DR)	+	++	+	+
Germany (FR)	++	++	+	+
Greece		+	+	+
Hungary	+	+		
Ireland	+	+		
Italy	++	++	+	++
Norway	++		++	
Poland	++	+		
Portugal		+	+	
Romania				+
Spain		++	+	
Sweden	++		+	++
Switzerland	++			
The Netherlands		++	++	+
United Kingdom	++	+		
USSR	+	++	+	+
Yugoslavia				+

\+ = identified problems.
++ = serious problems.

Table 5b. Eutrophication problems in continents others than Europe. Modified from Vollenweider (1979)

Geographic region and countries	Natural lakes	Reservoirs, rivers and irrigation systems	Estuaries, lagoons and closed seas	Marine coastal waters
North America				
Canada	++	+	+	
U.S.A.	++	++	++	+
Central America				
Mexico	+	++		
Guatemala/ Nicaragua	+			
Caribbeans	+	+		+
South America				
Venezuela/Surinam	+	+	++	
Columbia/Ecuador/ Peru	+	++	+	+
Brazil	+	++	++	+
Argentina/Chile	+	++	+	
Africa				
North	+	++	+	
Central	+	++	+	
South		++		
Asia				
India/Pakistan	+	++	+	
Indochina	+			
China	++	+	+	
Japan	++	+	++	+
Indonesia/ Philippines		++	+	
Australia & New Zealand	++	+	+	

\+ = identified problems.
++ = serious problems.

6.1 Lakes and reservoirs

Table 6 lists some of the more serious cases of eutrophied lakes and reservoirs. As a complete listing is not possible, an attempt had been made to evaluate the relative frequency of oligotrophic, mesotrophic and eutrophic lakes. To this end, Table 7 summarizes some statistics obtained from several studies. The various columns are not entirely independent, insofar as some lakes are counted twice or more. E.g., the data extracted by Lee and Jones

Table 6. Selected eutrophic lakes and reservoirs or bodies of waters which have been eutrophied above their natural state

1. Europe
   - Mjosa
   - Vattern
   - Malaren
   - L. Paajarvi
   - L. Esrom
   - Lough Neagh
   - L. Plon
   - Lac de Nantua
   - L. Ladoga
   - L. Balaton
   - L. Constance
   - L. Léman
   - L. Zurich
   - L. Baldegg
   - L. Sampach
   - L. Hallwil
   - L. Lugano
   - L. Como
   - L. Iseo
   - Wahnbach Reservoir

2. North America
   - L. Simcoe
   - L. Erie
   - L. Ontario
   - L. Memphremagog
   - L. Washington
   - L. Mendota
   - L. Tahoe

3. Central America
   - L. Cajititlan
   - L. Amatitlan
   - L. Valencia
   - L. Paranoa
   - L. San Roque Reservoir
   - L. Titicaca
   - Poza Honde Reservoir

4. Africa
   - L. Chad
   - L. Victoria
   - L. Tanganyika
   - L. Mariout
   - Kariba Reservoir
   - L. George
   - Hartbeespoort Reservoir

5. Asia
   - L. Kasimigaura
   - L. Biwa
   - L. Suva
   - Laguna de Bay
   - L. Songkhla
   - Doug Hu
   - Xi Hu
   - Xuanwu Hu
   - Mochou Nu
   - Mogu Hu

6. Australia
   - L. Burley Griffin

contain much of the OECD lakes, yet cover also lakes and reservoirs outside this programme. Conversely, the Canadian data reflect only partially the real situation of the country as they refer almost exclusively to its southern more inhabited belt, yet includes some of its most important water resources like the Laurentian Great Lakes. Therefore a simple counting of lakes can hardly do justice to the value of the resource.

Nonetheless, the listing provides insight into some aspects. Firstly, it reflects the difficulty of obtaining an objective picture from a but cursory though time-consuming survey. However, it also indicates that, indeed, there is large disparity between regions. E.g., the frequency distribution of the Canadian versus the U.S. data is consistent with the

Table 7. Eutrophication of lakes and reservoirs: % oligotrophic, mesotrophic and eutrophic (+hypertrophic) as classified in various comparative studies and reports

Region and/or countries	% oligo	% meso	% eu	Number of cases	Source of data evaluated
18 OECD countries (W. Europe, USA, Japan, Australia)	18	17	65	101	OECD 1982
OECD + Canada	48	16	36	230	Author
OECD + Canada + other countries	30	35	35	335	Jones, Lee 1986*
Canada	73	15	12	129	Janus, Vollenweider 1981
USA	7	23	70	493	Lorenzen 1979
Italy	29	28	43	65	IRSA 1980
Germany	8	38	54	72	LAWA 1985
Baltic Belt countries	15	38	31(+16)	130	Hillbricht-Ilkowska 1984**
Japan and other countries	25	39	36	36	Shiga '84 Data
China	44	32	24	34	Hongliang 1986
11 PAHO countries (South and Central America)	24	20	56	25	CEPIS, 1986
South African reservoirs	31	41	28	32	Thornton, Dalmsby 1982

*Total surface ca. 280,000 km^2 (22% of surface area of all lakes).

**The classification used by the author is not entirely corresponding to the trophic classification used here, yet sufficiently comparable.

reality of the two countries in the sense that of the 3/4 million of lakes in Canada the overriding majority is still oligotrophic while the lakes of the southern more sensely populated regions are prevalently eutrophic. Among the smaller USA lakes, there are many which are surrounded by farm land; their state, therefore, reflects eutrophication from agriculture and animal husbandry, yet may also in part be natural.

Systematic assessments of this sort are available only for a few countries. For Europe, the situation in Germany and Italy is comparable insofar as all important lakes (for Italy with the exception of the High Alpine lakes) have been evaluated with comparable methods. Among the 65 Italian lakes, 28 are identified as eutrophic; in Germany, out of the 72 lakes, 39 have been classified as eutrophic. Many of these lakes are reported to have been oligo- or mesotroph originally. However, the changing conditions in both directions (increasing eutrophication, and oligotrophication following initiation of remedial measures against eutrophication) make it at times difficult to classify lakes (or reservoirs) in a simple way. Austria, e.g. has summed up the trophic evolution of 28 lakes located in its territory. Some 2/3 of these lakes have had a history of first increasing and then decreasing eutrophication thanks to concerted remedial programmes. This applies also to some lakes of Switzerland, Germany, Sweden, Canada a.o.

In more dry climate regions, where water resources are often stored in artificial and semi-artificial impoundments, the situation remains quite precarious. E.g. from the some 800 reservoirs in Spain, at least 1/3 are highly eutrophied. Serious eutrophication problems in reservoirs are also reported from South America, South Africa, Australia, Mexico and elsewhere.

Besides the number of water bodies eutrophied, a further judgement element which needs be considered is the size of the bodies of water affected. Although a substantial number of the most valuable deep lakes have been severely eutrophied, the overriding majority of lakes and reservoirs in this category are relatively small in both surface area and volume as examplified with the OECD data; cf. Tables 8a and 8b. This is consistent with figures from China. From the 34 lakes assessed (representing some 24,300 km^2 of the 80,650 km^2 lake surface of China), 15 (44%) are classified as oligo- to mesotrophic, and 8 (23.5%) as eutrophic and hypertrophic. In terms of surface, however, the class of highly eutrophic lakes represents less than .5%, while the oligo- to mesotrophic lakes cover a surface of some 61%. The remaining 38.5% would be meso- to eutrophic.

Despite many uncertainties, the picture which evolves from this review indicates that, on a global scale, some 30 to 40% of the lakes and reservoirs are affected by more or less serious conditions of eutrophication. That many of these water bodies are relatively small, and hence in terms of total water mass, the percent figures may be smaller, is of but little importance. Economically speaking, many of these water bodies are extremely important either for drinking water supply, or for recreational purposes. Put into another way, the 525 km^3 of the lakes of the OECD programme listed as eutrophic would be sufficient to provide the water (calculated to 500 l/day/pc) for a population of 2.9 billion for one year. If one extends the figures given in Table 8, assuming the total volume of water stored in lakes to be 125,000 km^3, then about 37,000 km^3 would be mesotrophic, and some 3000 km^3 would be eutrophic.

Table 8a. Trophic conditions of 180 natural lakes (OECD + Canada study; excluding reservoirs)

	oligo	N	meso	N	eu	N	Total
Area (km)	144 470	71	101 201	43	32 600	66	278 331
%	51.9	39.4	36.4	23.9	11.7	36.7	100
Volume (km)	16 000	71	7 132	42	525	66	23 657
%	67.6	39.7	30.2	23.5	2.2	36.9	100
Average mean depth (m)	22.8		35.9		24.1		
Average water residence time (y)	15.8		6.0		3.1		

* Represents ca. 22% of world fresh water stored in lakes.

Table 8b. Trophic conditions of 34 reservoirs studied in the OECD programme

	oligo	N	meso	N	eu	N	Total
Area (km^2)	3.8	5	12.9	3	237.2	26	253.9
%	1.5	14.7	5.1	8.8	93.4	76.5	100
Volume (km^3)	67.2	5	187.2	3	2168.9	26	2433.3
%	2.8	14.7	7.7	8.8	89.5	76.5	100
Average mean depth (m)	14.1		14.2		8.1		
Average water residence time (y)	2.1		.3		1.0		

6.2 Rivers and canals

Although one can question the significance of the trophic classification applied to running waters, it is certain that many rivers today show productivity levels higher than natural. Eutrophication effects in rivers are generally less acute than in stagnant waters; some benthic productivity may even be beneficial in terms of self-purification capacity. Also, increase in potamoplankton is rarely considered a problem. Problems with potabilization of river water were recetly encountered in the River Nile following changes in its hydrological regime due to the construction of the High Dam at Aswan. This has created difficulties for the drinking water supply for Cairo and other sites. Eutrophication problems have been reported from impounded rivers like the Volga and Dnjepr systems in U.S.S.R. One of the most eutrophied rivers in Western Europe is the Loire in France which, in its lower part, reaches chlorophyl values as high as 500 mg/m^3. Reduction of water flow by macrophytes in canals has been reported from India, and interference by macrophytes with river navigation from Europe.

6.3 Estuaries, fjords, lagoons and coastal areas

In the marine environment, the waters that are readily subject to eutrophication are those that have a limited interaction with the pelagic waters adjacent to them. Yet even some unconfined coastlines have suffered considerable deterioration. Among the former, examples are the Oslo Fjord, Tokyo Bay, Manila Bay, Chesapeake Bay and the Potomac River, the Long Island Sound, the Bay of Fundy, the Strait of Juan de Fuca, the Lake of Tunis, Lake Maracaibo, the Saronikos Bay, Port Phillip Bay, Bay de Guanabara, Malaya Straits, the Seto Sea and the Java Sea; among the latter, some coastlines of Spain, France, Germany, the Baltic, the northern Adriatic Sea, the Black Sea, California, Florida, regions of Indonesian and Caribbean Islands, the mouth of the Amazonas, the Rio de la Plata a.o.

Even though the open sea cannot seriously be considered threatened by the eutrophication process, it must be remembered that closed seas are more susceptible. Reliable statistics on these situations are still lacking, since frequently the reports available refer simply to pollution from industrial and human wastes, without proper specification of alterations in productivity. The potential or actual damage in these zones is not just of an aesthetic nature, but often affects the fishery directly and human health indirectly. Beaches and/or fishing had to be closed temporarily in France, Spain, Italy, Florida a.o. Among the serious cases of coastal eutrophication, the precarious situation developed south of the River Po has called for large scale governmental programmes which reach far beyond the local community.

7. EUTROPHICATION CONTROL

7.1 Prediction

Control of eutrophication, in order to be successful, has to be based on sound scientific principles, and a thorough understanding of the cause-effect relationships. On such grounds, prediction becomes possible. Though the scientific understanding in many areas is still insufficient, in others substantial progress has been made. In particular, the ability to predict the behaviour of bodies of waters to changing nutrient load has improved considerably over the last two decades. Dynamic models are now in place which permit simulation of a variety of dynamic processes such as primary production, phytoplankton, and to some degree zooplankton biomass, and the oxygen regime. While such models call for a substantial pre-knowledge of the specific properties and characteristics of the waters in question and, hence, a considerable amount of data (Jorgensen, 1980), they are built on a relatively simple mass balance conception which requires that the

Change of nutrient in the water body = External supply - Export loss ± Exchange with sediments

Further splitup of these equations into process equations (which consider morphometry, hydrodynamics, energy input, chemical and biological processes) yield the basis for more precise simulations.

One of the simplest equation of this sort for e.g. phosphorus can be written as

$$\frac{dP^*}{dt} = P_i^* - rP^* - sP^*$$

P^* = total mass of phosphorus in the lake (kg/y)
P_i = external supply (kg/y)
r = flushing coefficient (1/y)
s = sedimentation coefficient (1/y)

Modification and development of this equation permits to estimate the loading tolerance (critical loading) of bodies of waters which is defined as the level of loading when exceeded leads to eutrophication. Knowledge of the critical loading is of importance for both, prevention and cure of eutrophication. From the management point of view, a correct evaluation of the loading tolerance and/or the nutrient reduction required is further of utmost economic importance. Incorrect estimates can lead to substantial economic losses. However, it must be noted that nutrient load control, though the most important measure to reduce eutrophication, cannot always be done to the extent necessary.

7.2 Single case technologies and interventions

A large array of technologies and more or less direct measures have been developed over the last decades to combat eutrophication. Nutrient load control is the most desirable way to reduce eutrophication, yet experience with other forms of intervention is growing daily.

The following listing provides an overview of the most common measures and potential interventions.

A. Measures outside the body of water:

(1) treatment of wastewaters (elimination of phosphorus and nitrogen);
(2) complete diversion of the wastewaters (ring calanizations);
(3) primary sedimentation basins;

(4) direct precipitation of nutritive substances in the affluents;
(5) watershed protection (reforestation; restriction of the use of fertilizer; restriction of industrial livestock breeding establishments; controlled fert-irrigation, etc.;

B. Corrective measures inside the body of water:

(1) physical manipulation (destratification; hypolimnetic aeration; withdrawal of hypolimnetic water; alteration of flushing regime);
(2) chemical and sedimentary manipulation (nutrient precipitation inside the body of water; inactivation and removal of sediments);
(3) biological manipulation (mechanical harvesting of the biomass; macrophytes, algae, fish, application of toxic substances; herbicides, algicides, pesticides; direct manipulation of the food chain and of the biological equilibrium).

Examples of each type of intervention, applied singly, or in combination, are available in the scientific and technological literature. Some of the applied measures have been very successful, others have been partially successful, but also examples of complete failure are known. Based on the positive experiences, we can say with certainty that reversal of the eutrophication process is indeed possible. However, the less successful examples indicate that our knowledge is not complete. Critical analysis of such cases is necessary to understand the various "whys" for failures. The choice of corrective or preventive measures has to be based not only on general knowledge of the biological and physical dynamics of bodies of water, but also has to be tuned to the specific properties of the body of water in question. Hypolimnetic aeration, for instance, is not a suitable measure for either very deep or very shallow lakes, but can often be successfully applied to bodies of water of intermediate depth. Nutrient diversion by means of ring collectors can be very effective in protecting a lake but at the same time can be a threat to downstream waters. Manipulation of the foodweb by introducing new fish species may control e.g. aquatic weed problems but can also have effects on the intraspecific balance of the fish communities, etc.

8. NEED FOR COMPREHENSIVE STRATEGIES AND POLICIES TO CONTROL EUTROPHICATION

The process of "accelerated" eutrophication is essentially caused by three elements which are interrelated and directly linked to worldwide demographic changes:

(a) rapid general increase in population with a strong tendency toward urbanization and resulting rapid increase in urban waste discharged directly into the waterways, lakes and coastal areas. An added factor to this since the end of World War II is detergents containing polyphosphates;

(b) rapid industrialization linked to population growth with the corresponding increase in industrial waste of all kinds, some of them containing the nutrients needed for the growth of algae and macrophytes;

(c) intensification of agriculture and changes in production methods, through preferential development of monocultures, increased use of chemical fertilizers, concentration of livestock breeding, direct discharge of agricultural wastes into the waterways, etc.

The relationship between the demographic development with all its ramifications and the extent of eutrophication can be demonstrated with many examples. E.G. the patterns resulting from mapping of factors useful in quantifying the degree of eutrophication (such as chlorophyll or primary production) for the Laurentian Great Lakes show striking complementarity to the maps of human population density of the corresponding catchment basins. Similarly, areas mostly affected by eutrophication in the Mediterranean coincide with densely populated interlands, which also are areas of intensive agriculture and high industrial development. Also, in Japan the areas with the highest incidence of eutrophication are adjacent to areas of high population density. Coincidences of this sort could easily be extended to other parts of the globe.

What evolves from this points to the need to pursue the problem of eutrophication, not only in terms of more o less disconnected, single-case solutions, but also in terms of regional and supra-regional management approaches. In other words, in order to bring eutrophication under firm and lasting control, the more or less "ad hoc" designed technical, single-case interventions need to be sustained by supplementary comprehensive medium-term and long-term strategies and policies.

These are to include appropriate territorial planning, aimed at harmonizing urban, industrial and agricultural developments. These strategies demand a thorough analysis of demographic evolution, on both regional and national, and sometimes on international levels. It is evident that at the national level this implies a considerable economic, administrative and legislative commitment guided by a coherent social policy. Seen in this light, the eutrophication problem automatically becomes part of the more general problem of cleaning up and protecting the environment. Such a direction is already being taken in many technologically and economically developed countries. But precisely in these countries, which are rapidly reaching the limits of demographic expansion, options for comprehensive long-term strategies are more limited than in developing countries, at least in principle.

Among the countries which have included the problem of eutrophication in their legislative programs for cleaning waters are Sweden, Denmark, Germany, Austria, Switzerland, Italy a.o. In Sweden, 80% of this country's treatment plants include a third stage for the elimination of phosphorus; only 20% of the waters discharged into the waterways receive no treatment. Under the 1972 Treaty between Canada and the USA, approximately 75% of the waters discharged into the basin of the Great Lakes receive primary and secondary treatment defined as "adequate".

The reduction of polyphosphates in detergents (imposed by law, e.g. in Canada and some U.S.A. states bordering on the Great Lakes, Switzerland, Italy a.o.) can be viewed as a medium-term strategy which, however, cannot be regarded on its own as a substitute of a more comprehensive one, but should be part of it.

Strategies for rationalization of some agricultural sectors under an environmental point of view are only at the beginning stage. Aggravated by the more or less general crisis in agriculture, independent of the social, economic and political context, the problems arising in this regard are enormous and cannot be overcome by legislation only. Controlled use of artificial fertilizers would seem to be the easiest sector to enforce, yet, depending on local conditions, might remain but a partial measure. In Denmark, where agricultural contribution to eutrophication of lakes and coastal areas is particularly high, the government has introduced a computerized and indirect control system of the use of fertilizers. It is left to the farming community to reduce the amount of fertilizers, yet, with the provision that--if the desired level is not reached within a stated time frame--fertilizers would be highly taxed.

The loss of nutritive substances to waterways as a result of intensive livestock breeding is in many places far greater than the loss due to the use of artificial fertilizers in crop fields. Better utilization of wastes, either through proper recycling and/or a more rational transformation into products compatible with the environment, or as a source of energy, however marginal, imposes itself as an unavoidable step toward a lasting solution. The regional government of Emilia-Romagna, Italy has introduced legislation which imposes a ceiling for hogs per available land surface; other legislation provides incentives to farmers for improved utilization of farm waste.

In the industrial sector, primary industries related to transformation of agricultural products, and industries producing fertilizers, as well as a few others, are among those contributing to eutrophication by discharging nutrient-rich wastes to waterways. The direct contribution to the problem by other industries--which may be the main offenders in terms of water quality--is less important, though perhaps not negligible if one takes into consideration that industrial wastes often contain micro-elements that promote plant growth. Therefore, in a comprehensive planning scheme, location or relocation of some industrial complexes may also have to be considered in terms of beneficial effects on reducing eutrophication.

While it is difficult to make appropriate economic evaluations, the figures provided by Austria at least give some ideas of the costs of eutrophication. The investment figures for the period 1989-1995 for the 28 lakes (covering some 960 km^2) amount to 9220 mill. shilling (ca 740 mill. U.S. $) which means that sanitation of these lakes costs Austria some 1.3 million U.S. $/$km^2$. In Switzerland, the installation costs for two lakes (L. Baldgg, 5.25 km^2, and L. Hallwil, 9.95 km^2) are estimated to 3.6 mill. SFr (ca. 2.5 mill. U.S. $) with running costs of 730,000 SFr/y (ca. .5 mill. U.S. $).

SELECTED BIBLIOGRAPHY SOURCES

AUBERT, M. & J. AUBERT (1986) Eutrophie et dystrophie en milieu marin. Rev. Intern. d'Océanographie Médical., Tomes 83, 84.

BARICA, J. (1987) Water quality problems associated with high productivity of Prairie lakes: A review. WHO/Water Qual. Bull. 12(3).

BAXTER, R.M. (1977) Environmental effects of dams and impoundments. Ann. Rev. Ecol. Syst. 8.

BERNHARDT, H. (Ed) (1978) Phosphor. Weg und Verbleib in der Bundesrepublik Deutschland. Verlag Chemie. Weinheim, New York.

BMLF (Ed. H. Sample) (1982) Seenreinhaltung in Osterreich. Bundesministerium fur Land und Forstwirtschaft (BMLF). Wien, 1982.

CULLEN, P. R.S. ROSICH & P. BEK (1977) A phosphorus budget for Lake Burley Griffin and management implications for urban lakes. The Australian Water Resources Council: Report 75/92.

DEGENS, E.T. (Ed) (1982) Transport of Carbon and Minerals in Major World Rivers, Part 1 to 3. Geol.-Palaont. Inst. Univ. Hamburg, Hamburg, 1982.

FEACHEM, R.G. (1977) Infectious disease related to water supply and excreta disposal facilities. Ambio 6, 1.

FLANAGAN, P.J. & P.F. TONER (1975) A preliminary survey of Irish lakes. An Forbartha Publ., Dublin.

HILLBRICHT-ILKOWSKA, A.G. (1984) The indices and parameters useful in the evaluation of water quality and the ecological state of temperate lowland lakes connected with their eutrophication. Proc. Shiga Conference '84: Conservation and Management of World Lake Environment (LECS '84).

HONGLIANG, L. (1986) A study on the present situation of eutrophication of lakes and its prevention and control in China. Chinese Res. Acad. Environm. Sciences, Beijing. Manuscript.

IJC (Berg, A.N. & M.G. Johnson) (1978) Environmental Management Strategy for the Great Lakes System. International Reference Group on Great Lakes Pollution from Land Use Activities (PLUARG), International Joint Commission, Windsor.

IRSA (Chiaudani, G., M. Gerletti, R. Marchetti, A. Provini & M. Vighi) (1978) Il problema dell' eutrofizzazione in Italia. Istituto di Ricerca sulle Acque. Consiglio Nazionale delle Ricerche, No. 42, Roma.

IRSA (Passino, R.) (1980) Indagine sulla qualita delle acque lacustri italiane. Istituto di Ricerca sulle Acque. Cousiglio Nazionale delle Ricerche, No. 43, Roma.

JANUS, L.L. & R.A. VOLLENWEIDER (1981) The OECD Cooperative Programme on Eutrophication. Canadian Contribution, NWRI, Canada Centre for Inland Waters, Scient. Series 131.

JONES, R.A. & G.F. LEE (1986) Eutrophication modelling for water management: An update of the Vollenweider-OECD model. WHO/Water Qual. Bull. 11(2).

JORGENSEN, S.E. (1980). Lake Management. Water Development, Supply and Management: 14, Pergamon Press.

LAWA (Landergemeinschaft Wasser) (1985) Seen in der Bundesrepublik Deutschland. Druck Bayrisches Staatsministerium des Innern.

LECS' 84. Data Book of World Lakes. Shiga Conference '84 on Conservation and Management of World Lake Environment. OTSU 1984.

LORENZEN, M.W. (1979) Effect of phosphorus control options on lake water quality. Report EPA-560/11-79-011.

LUND, J.W.G. (1980) Eutrophication in the United Kingdom. The Soap and Detergent Industry Assoc. Manuscript.

MEYBECK, M. (1982) Carbon, nitrogen, and phosphorus transport by world rivers. Am. J. of Science, 282.

MMBW (Croxford, H.) (1973) Environmental Study of the Port Phillip Bay. Melbourne and Metropolitan Board of Works, and Fisheries and Wildlife, Dept. of Victoria, Melbourne, Australia.

MOPU (Ortiz Casas, J.L., R. Pena Martinez, F.G. Lee & R.A. Jones) (1983) Aportacion de nutrientes y eutrofizacion de embalses. Centro de Estudios y Experimentacion de Obras Publicas, y Centro de Estudios Hydrograficos, Madrid.

McCOLL, R.H.S. & H.R. HUGHES (1981) The effects of land use on water quality--A review. Nat. Water & Soil Conserv. Org., Wellington: No. 23.

OBENG, L. (1977) Should dams be built? The Volta Lake Example. Ambio, 6, 1.

OECD (Vollenweider, R.A. & J.J. Kerekes) (1982) Eutrophication of Waters. Monitoring, assessment and control. OECD, Paris.

PAHO-CEPIS (Salas H.J. & G. Limon) (1986). Memoria del tercer encuentro del proyecto regional: Desarrollo de metodologias simplificados para la evaluacion de eutroficacion en lagos calidos tropicales. Manuscript.

REGIONE EMILIA-ROMAGNA (R. Marchetti) (1984) Indagini sul problema dell'eutrofizzazione delle acque costiere dell' Emilia-Romagna. Assessorato Ambiente e Difesa del Suolo, Bologna.

ROSENBERG, R. (1985) Eutrophication--the future marine coastal nuisance. Marine Poll. Bulletin, 16.6.

SNV (Ahl, T. & T. Wiederholm) (1977) Svenska Vatten-kvalitets Kriterier: Eutrofierande Amnen. Statens Naturvardsverk SNV-PM 918. Uppsala.

SZKIELDA, K.H. (1982) Investigations with satelites on eutrophication of coastal regions. Mitt. Geol.-Palaont. Inst. Univ. Hamburg, SCOPE-UNEP Sonderband H.52.

TALLING, J.F. (1965) The chemical composition of African lake waters. Int. Revue ges. Hydrobiol. 50.3.

THORNTON, J.A. & R.D. DALMSBY (1982) Applicability of phosphorus budget models to Southern African man-made lakes. Hydrobiology 89.

TSUJITA, T. (1984) Sequential processes in the occurrence of blooms and red tide in the sea. In: Progrès récent dans les sciences de la mer. Soc. Franco-Japonaise d'océanographie, Tokyo, Japan.

TUNDISI, J.G. (1981) Shallow waters in South America: Present knowledge and perspectives for future management. Proc. SWPE/UNEP Workshop "Dynamics of Continental Wetlands and Water Bodies, URSS, 1981.

VIGHI, M. & G. CHIAUDARI (1986) Eutrophication in Europe. The role of agricultural activities. Rev. Environm. Toxicology.

VOLLENWEIDER, R.A. (1968) Scientific fundamentals of the eutrophication of lakes and flowing waters, with special reference to nitrogen and phosphorus as factors in eutrophication. OECD DAS/CSI/68.27, Paris.

VOLLENWEIDER, R.A. (1981) Eutrophication: A global problem. WHO/Water Qual. Bull. 6(3).

WHITE, E. 1983. Lake eutrophication in New Zealand: A comparison with other countries of the OECD. New Zealand J. Marine & Freshw. Res. 17.

WHO (L. Landner) (1976). Eutrophication of Lakes: Causes, effects and means for control. WHO Regional Office for Europe.

WHO-GEMS/WATER DATA EVALUATION REPORT, WHO, Geneva, 1983.

WHO/WATER QUALITY BULLETIN (1977-1987, Vol. 1 to 12) S. Barabas (Ed), CCIW, Burlington, Ont., Canada.

WRC (Walmsley, R.D. & M. Butty) (1980) Guidelines for the control of eutrophication in South Africa. Water Research Commission, National Institute for Water Research, Pretoria.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

RECOVERY OF THE LAURENTIAN GREAT LAKES, 1970-1985: EUTROPHICATION ASPECTS

J. BARICA

Lakes Research Branch, National Water Research Institute,
Canada Centre for Inland Waters
Burlington, Ontario, Canada

Abstract

Eutrophication of the Laurentian Great Lakes accelerated exponentially during the first half of this century. Canada and the United States recognized this fact and signed the 1972 Great Lakes Water Quality Agreement. Since then, coordinated activities have resulted in a massive cleanup, unprecedented in history. Major wastewater treatment programs have been implemented. Industrial controls and phosphate limitations in detergents produced a reduction and even a reversal of the eutrophication process. The long-term trend of total phosphorus concentrations have been downward since the mid 70's when legislation on phosphorus reduction was put into effect. Nuisance algal blooms are no longer a common occurrence and Cladophora infestations of nearshore areas are declining.

Concurrently, with the success in phosphorus control, new phenomena were identified, namely, increases in nitrates, N:P ratios, taste and odor problems and interactions between the nutrients and toxic contaminants. This overview highlights characteristic examples of the recent reversals in long-term trends of some water quality parameters related to eutrophication. It also identifies historic peak periods of nutrient pollution and discusses future projections and lake rehabilitation strategies with the focus on near-shore Areas of Concern.

INTRODUCTION

The Laurentian Great Lakes, with a drainage basin comparable in size to Central Europe, contain about twenty percent of all freshwater on the Earth's surface. They represent a major natural resource for both Canada and the

United States. Thirty percent of Canada's population (7.5 million people) and twenty percent of the United States population (40 million people) live in the Great Lakes drainage basin. Twenty-four million people depend on the Great Lakes for drinking water supply. Considerable industrial development (e.g. 50% of U.S. steel production; 62% of Canadian steel production) has occurred within the basin because of the availability of abundant, inexpensive water and accessible, efficient transportation (Thomas and Hartig, 1988).

Since the days of early extensive settlement in the late 1880's accompanied by agricultural and industrial development, the population has been increasing exponentially in the basins of Lake Erie and Ontario (Dobson, 1981). This development led to human mismanagement and accelerated deterioration of water quality, recognized soon by both countries. After several epidemics of cholera and typhoid fever - caused by discharges of human waste - the Boundary Water Treaty was signed by the United States and Canada in 1909 and established the International Joint Commission as a unique binational organization seeking to reach consensus on solutions to common problems, including water and air pollution, lake levels, power generation, and other issues of mutual concern. Industrial growth was followed by oil pollution (1940s) and accelerated eutrophication in the 1960s, as a result of uncontrolled input of nutrients from municipal sewage containing detergents, industrial wastes and agricultural runoff. By the late 1960s the degradation had become extreme in some areas, primarily the Western and Central Basins of Lake Erie, Green Bay in Lake Michigan, Saginaw Bay in Lake Huron, Hamilton Harbour in Lake Ontario and others. Massive algal blooms were occurring frequently and a number of municipal and industrial areas were devoid of visible aquatic life (Forde, 1969). There were massive die-offs of fish (mostly "alewives") in Lakes Michigan and Ontario, severe oxygen depletion in hypolimnetic waters of Central Lake Erie and deterioration of fisheries. At the same time, the 1960s also saw a dramatic increase in public concern about the degraded state of these areas, with headlines like "Is Lake Erie dead?" appearing in the news.

In response to this catastrophic situation, the International Joint Commission initiated a review of the state of the lower Great Lakes (Erie and Ontario). The results were released in 1969 and marked the beginning of official action, unprecedented so far in history, with the objective to stop further degradation and initiate restoration.

This review, prepared for the international audience of the 3rd International ILEC Conference "Balaton 88", summarizes major achievements of the concentrated remedial effort in both countries over the period of the past 15 years with the intention to demonstrate an example to be followed in the basins of the world's other large lakes.

PHOSPHORUS CONTROL PROGRAMMES IN THE GREAT LAKES BASIN.

Despite a strong controversy about the key nutrient controlling eutrophication process (Legge and Dingeldein, 1970) and thanks to dedicated effort of some individuals and groups (Vallentyne 1970, a,b) it was finally accepted that it is phosphorus that is the limiting and at the same time the only controllable nutrient. Analysis of phosphorus sources revealed that inputs were chiefly from laundry detergents, used in households, human waste and agricultural runoff (Chapra, 1977). These three major sources were chosen as the primary targets for coordinated reduction of P-loadings, in the 1972 Great Lakes Water Quality Agreement (GLWQA), signed by both countries. Specific tables and goals for annual phosphorus loading reductions were developed for both countries and each lake (GLWQA, 1973), (i.e., for Lake Erie, total reduction between 1972 and 1976 was set for about fifty percent, from 32,000 tons per year to 16,100 tons in 1976; for Lake Ontario, from 18,700 to 10,000 tons per year).

Overall long-term objectives of the phosphorus reduction programs was to minimize eutrophication problems in the Great Lakes. It was anticipated that successful implementation of these programs would accomplish the following results:

(a) Restoration of year-round aerobic conditions in the bottom waters of the central basin of Lake Erie;

(b) Reduction in present levels of algal growth in Lake Erie;

(c) Reduction in present levels of algal growth in Lake Ontario, including the International Section of the St. Lawrence River;

(d) Stabilization of Lake Superior and Lake Huron in their present oligotrophic state.

The Parties, in cooperation with the State and Provincial Governments, and with the International Joint Commission, committed themselves to monitor the extent of eutrophication in the Great Lakes and the progress being made in reducing or preventing it.

Gradual implementation of these programs took several years and was completed in principle around 1973-1976, with some additional improvements stipulated in 1978 and 1984 revised GLWQ agreements. Phosphate ban legislations for laundry detergents was accepted in Ontario and most of the U.S. states, except Ohio and Pennsylvania (Ohio introduced the legislation only in 1988). Upgrading of sewage treatment plants to increase chemical removal of phosphorus and significant improvements of agricultural wastes management and land practices were instrumental in the P-reduction programs: the total cost is estimated to be 20 billion dollars.

Specific effluent requirements were established (i.e. daily average of 1 mg/L of P in municipal wastewater after treatment), as well as timetables for gradual reduction of phosphate levels in laundry detergents from pre-agreement levels of 30-40 % as P_2O_5 down to 5% (The 20 % limit was introduced before the signing of the Agreement, in summer of 1970 (Bruce and Higgins, 1977). The funds for these major operations were provided by both governments in the form of loans and grants to municipalities, the agricultural sector, industries and research.

LONG-TERM RESULTS OF PHOSPHORUS CONTROLS - ANALYSIS OF LONG TERM TRENDS.

Phosphorus loadings.

As a result of the successful implementation of measures introduced during late the early 70's as expressed in the 1972 GLWQA, the P loadings, particularly from municipal sources, dropped dramatically (Table 1) and by early 80's approached the target loadings established by the 1973 and 1978 Agreements. The loading reductions over the period were most pronounced in Lake Erie and Lake Ontario (approx. 80%). In the Upper Great Lakes the corresponding reductions were over 50% (Table 1).

Phosphorus concentrations and algal biomass.

Phosphorus loading reductions were clearly reflected in corresponding open lake concentrations of both total P (TP) and soluble reactive phosphorus (SRP) over the study period. The data are presented in Figure 1 together with chlorophyll a values as a measure of the phytoplankton biomass.

While total phosphorus long-term trends show a definite decline corresponding to reductions of P loads, chlorophyll a decreases are not so straightforward. This is due to the fact that the open waters of the Great Lakes, for which the trends are presented had relatively low starting concentrations of chlorophyll a and analytical "noise". While the trends presented in Figure 1 are statistically significant (IJC, 1987), SRP spring data for Lake Ontario (Dobson, 1984 and pers. comm.) present an even more convincing picture, with SRP values peaking during early 70, and steadily decreasing since (Figure 2). At the same time, Cladophora tissue phosphorus content decreased significantly over the period (Painter and Kamaitis, 1985, Fig.3).

Figures 1-2 were all based on spring concentrations. Kwiatkowski's (1982 and 1984) analysis of these parameters, considering their annual mean values also presents significant trends, but not as clearly visible as the spring concentration.

Table 1. Reported municipal phosphorus loadings in the Great Lakes Basins (tonnes per year). From IJC (1987)

LAKE BASIN	1972 ESTIMATE	1975	1976	1977	1978	1980	1981	1982	1983	1984	1985	% REDUCTION 72(75)-85
SUPERIOR												
United States		224	222	154	142	94	70	64	85	89	94	58.0
Canada		62	71	108	97	109	100	115	68	58	53	14.5
MICHIGAN												
United States		2,325	2,336	1,660	1,314	1,047	934	885	928	919	894	61.5
HURON												
United States		414	370	340	273	232	244	248	219	205	201	51.4
Canada		210	208	217	222	195	231	168	216	243	261	inc. 24.2
ERIE												
United States	13,870	6,719	5,578	6,147	5,250	3,287	2,642	2,199	2,356	2,499	2,176	84.3
Canada	1,390	232	262	259	228	213	232	250	274	268	273	80.3
ONTARIO												
United States	4,750	1,847	1,815	2,089	1,761	1,535	1,194	1,166	1,055	872	772	83.7
Canada	5,110	2,373	1,267	1,000	967	977	1,014	949	906	952	938	81.6

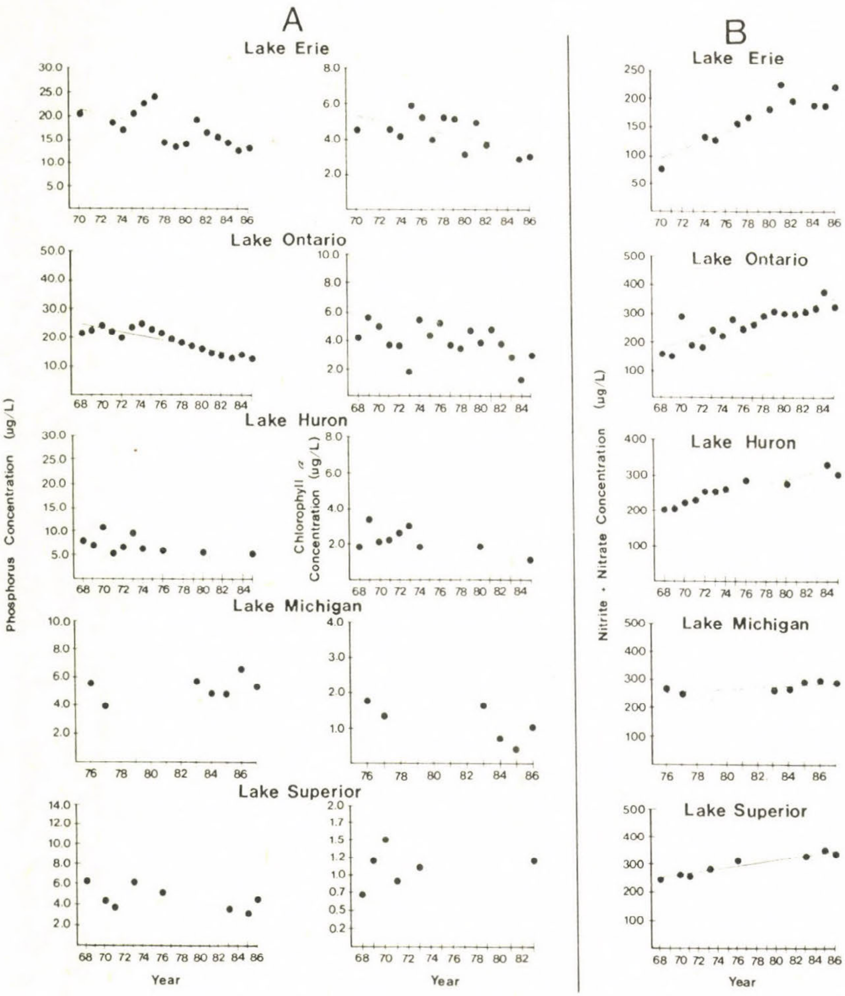


Fig. 1. Long term trend in total phosphorus, chlorophyll a and nitrite plus nitrate nitrogen concentrations in the Great Lakes (from IJC, 1987).

Perhaps even more convincing are the data on phytoplankton biomass from the near-shore areas where eutrophication has been historically more advanced. Bird and Rapport presented examples from two such areas in Bay of Quinte in Lake Ontario (Fig.4) demonstrating a significant reduction of phytoplankton biomass, and from Saginaw Bay in Lake Huron with altered phytoplankton composition and reduction of Cyanophytes as a

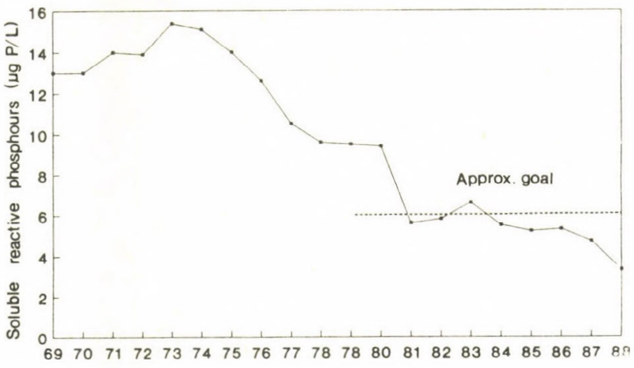


Fig. 2. Lake Ontario. Soluble reactive phosphorus in offshore, near-surface waters during March and April, 1969-1987. Great Lakes Surveillance dat CCIW, Burlington. Courtesy of H. Dobson, NWRI.

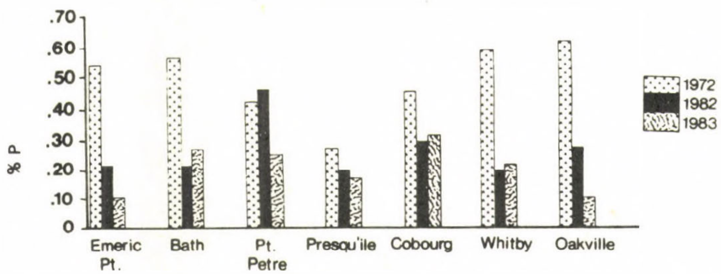


Fig. 3. Tissue phosphorus content of Cladophora before (1972) and after phosphorus controls (1982-1983). From Painter and Kamaitis, 1985.

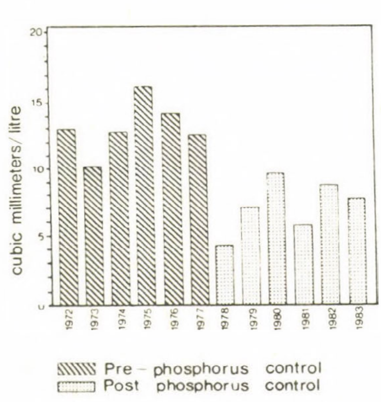


Fig. 4.

Concentration of phytoplankton in Bay of Quinte (Lake Ontario), before and after phosphorus controls, 1972-1983. From Bird and Rapport 1986.

result of P controls (Fig.5). Munawar and Munawar (1986) noted significant shifts in algal composition in the open water of the Great Lakes as well.

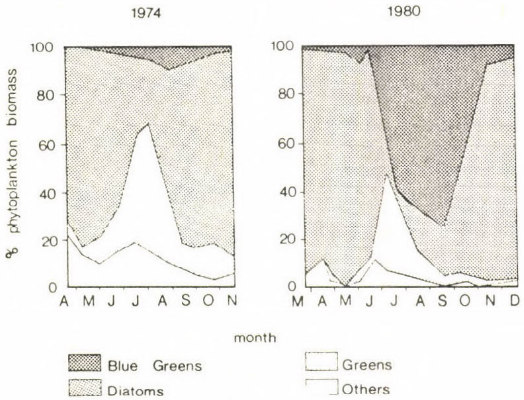


Fig. 5.

Concentration of phytoplankton crop composition before and after phosphorus controls in Saginaw Bay (1974 and 1980). From Bird and Rapport 1986.

It can be concluded that the P controls were successful and met in principle the expectations, except for some remaining problems in near-shore areas, as discussed later.

NEW PHENOMENA AND DEVELOPMENTS.

The Great Lakes Water Quality Agreement of 1972 generally reflected a state of knowledge in the 60s. A number of new water quality related issues have surfaced over the following decade. The most significant of them is the toxic contaminants problem (metals and organics) which has become the highest priority issue in the basin (treated separately by R.J. Allan in these proceedings) and incorporated in the 1978 update of the Great Lakes Water Quality Agreement.

Besides the toxics issue, there has been some other unexpected and noteworthy developments:

Slow Lake Erie hypolimnetic oxygen recovery.

It was assumed that P controls would have an immediate impact on improvement of hypolimnetic oxygen conditions in Lake Erie. One of the GLWQA's objectives was a year-round restoration of oxygen conditions. However, the fact is that the hypolimnetic oxygen depletion rates continued to increase over the study period (Figure 6). This controversy was subjected to a separate analysis (Barica 1981) concluding that the effect of P controls will take more time to be clearly noticeable than expected due to morphometry and internal loading of P in Lake Erie (Charlton, 1980; Rosa and Burns, 1987).

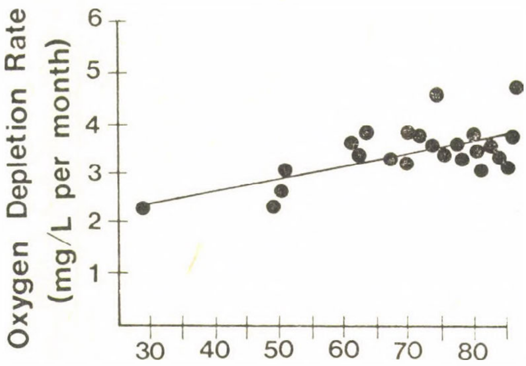


Fig. 6. Hypolimnetic oxygen depletion rates in Lake Erie, 1930 - 1985. From IJC 1987.

Increases of nitrate concentrations and N:P ratios.

Concurrently, with the success in phosphorus control, a new phenomenon became noticeable: the nitrate concentrations in the lake water increased alarmingly, exceeding levels of the 60's several fold. Dobson (1981) presented trends of springtime nitrate in Lake Superior, Lake Huron and Georgian Bay, Lake Ontario and Eastern Lake Erie, between 1960 and 1980, based on extensive data from Great Lakes water quality surveillance programmes. While the concentrations were still low from the health and drinking water quality point of view

($0.3 - 0.4$ mg/L NO_2-NO_3-N, as compared to 10 mg/L for drinking water; Health & Welfare Canada, 1978), the relative increases over the past twenty years were between 30 to over 200%, with highest percentage increases in the most populated and agriculturally productive basins of Lake Ontario and Lake Erie. The recent trends from IJC data were presented in Figure 1. It should be kept in mind that these data come from open lake sampling sites. However, the Great Lakes abound with numerous large and small nearshore embayments, with limited water exchange with the main lakes. In Hamilton Harbour, which is such an embayment, the nitrate increases have been even more dramatic. Between 1948 and 1979, the average annual concentrations of nitrate in Hamilton Harbour increased six-fold (Forde 1979). Significant increases have been observed also in the Bay of Quinte, Lake Ontario (Robinson 1986) and Severn Sound, in Georgian Bay on Lake Huron (MOE Remedial Action Plan Report, 1987).

Despite these increases, the lake water itself and its quality does not seem to be directly affected by nitrate or to be in danger for the foreseeable period of time (Concentrations less than 1 mg/L of nitrite and nitrate nitrogen together, as measured in Great Lakes Surveillance programmes).

Impact on lake water quality is so far indirect, mainly through alteration of N:P ratios. Low ratios are known to govern phytoplankton succession, particularly the onset of blue-green algae and formation of algal blooms. The low values (5:1 or less, Allan and Kenney, 1980) are characteristic of eutrophic systems, while values over 30:1 are typical of oligotrophic lakes (Forsberg 1979). Increases of nitrate raise total dissolved nitrogen levels and thus N:P ratios. This is even more pronounced if P values decrease concurrently during the same period. This has been the case of the Great Lakes since introduction of phosphorus control legislation in Canada and the U.S. in the early seventies. Stevens and Neilson (1987) estimated that the total phosphorus (TP) loading to Lake Ontario has declined from 14,000 t/yr in 1969 to 8,900 t/yr in 1982, with corresponding TP mid-lake spring concentrations decreasing from a maximum of 31 ug/L/yr in 1973 to 13 ug/L/yr in 1982. During the same period, spring nitrate (measured together with nitrite) has increased at a rate of 9.5 ug/L/yr causing N:P ratios to increase from 10 to 32.

N:P ratio increases have been reported from the Bay of Quinte (up to 24.4, Robinson 1986) and Severn Sound (threefold increases between 1969 and 1986 - up to 37.8 from 12.0; MOE Remedial Action Plan Report 1988). In all these cases, the increases of N:P ratios were due to a combined effect of nitrate increases and simultaneous phosphorus decreases by P control measures (upgrading of sewage treatment plants by chemical P-removal). This is a relatively beneficial phenomenon causing an "oligotrophicating" effect, and further improvement of Lake Ontario's trophic state, provided that P control measures are maintained. Relaxation of phosphorus controls would allow catastrophic eutrophication when such a large pool of nitrogen is already present.

An analysis of possible causes and impacts of nitrifying the Great Lakes was presented (Barica, 1987). It was concluded that the increases of nitrate in the Great Lakes are a combined result of several factors of differing relative importance. In the lower Great Lakes, the increased human population, increased use of fertilizers in agriculture and large-scale phosphorus reductions are predominant. In the upper Great Lakes (Huron, Superior) atmospheric deposition of nitrate in acid rain is the most important factor.

Taste and odour problem.

Since the mid 70's, instances of taste and odour incidents in drinking water supplies in some Great Lakes communities supplied with Lake Ontario water became more frequent than in the past.

The two most frequent causes of taste and odour in lake water are geosmin (trans, trans-1, 10-dimethyl-9-decalol) and 2-methylisoborneol (MIB, 1, 2, 7, 7-tetramethyl-exo-bicyclo[2.2.1]heptan-2-ol). They are produced by actinomycetes and Cyanophytes. Under certain conditions, actinomycetes can grow on decaying *Cladophora* to produce tastes and odours (Persson 1983).

In the summer of 1983, geosmin was identified in a municipal water supply drawn from western Lake Ontario. The geosmin concentrations were 0.01-0.07 ug/L, within the range for threshold odour concentration of 0.01-0.2 ug/L. 2-methylisoborneol was not detected. The odour 'event' coincided with a dieoff of Cladophora in the lake, but a direct link between the dieoff and geosmin production was not established. Decomposing Cladophora in shoreline areas produced a strong odour in the air. 3-Methylindole, elemental sulfur, dimethyl tetrasulfide, and dimethyl pentasulfide were tentatively identified in water samples collected from these areas, but geosmin and 2-methylisoborneol were not detected (Brownlee et al., 1984). The cause of increased incidence has not yet been identified.

Nutrient - contaminants interactions.

Over the past years it was observed that the fish from Lake Ontario are more contaminated than those from Lake Erie (Bird and Rapport, 1986). Loadings of contaminants in both lakes are fairly comparable. It was speculated that the contaminants in a more advanced eutrophic system become masked or removed by sedimentation within the food chain and do not reach the fish. Some preliminary observations from other lakes in the Great Lakes basins indicate similar conclusions (J. Carey, pers. comm.). Research is underway to elucidate this phenomenon and to assess water quality management implications (Allan, this issue).

STRATEGIES FOR THE 80'S AND 90'S; REMEDIAL ACTION PLANS

Stopping and even reversing the eutrophication process in the Great Lakes is probably the greatest large-scale environmental success in history. The total cost is estimated around 20 billion and there are no questions as to whether or not this investment was worthwhile. The Great Lakes are positively of better quality now than 15-20 years ago - this applies to the contaminants isssue also (Allan, this issue).

However, the improvements so far have been documented for open waters of the Great Lakes, since these were the primary objective of the U.S.-Canada Water Quality Agreements, based on the Boundary Treaty of 1909. This discrepancy was noted in 1985, when the IJC - after realizing that some, mostly near-shore areas - do not present visible improvements and a number of serious problems remain. The IJC has identified 42 Areas of Concern in the Great Lakes basin (Fig.7). In each of these areas, GLWQA objectives or jurisdictional standards, criteria or guidelines established to protect uses have been exceeded and remedial measures are necessary to restore beneficial uses, such as municipal and industrial water supplies, recreation and aquatic life. Areas of Concern include the major municipal and industrial centers on Great Lakes rivers, harbours and connecting channels.

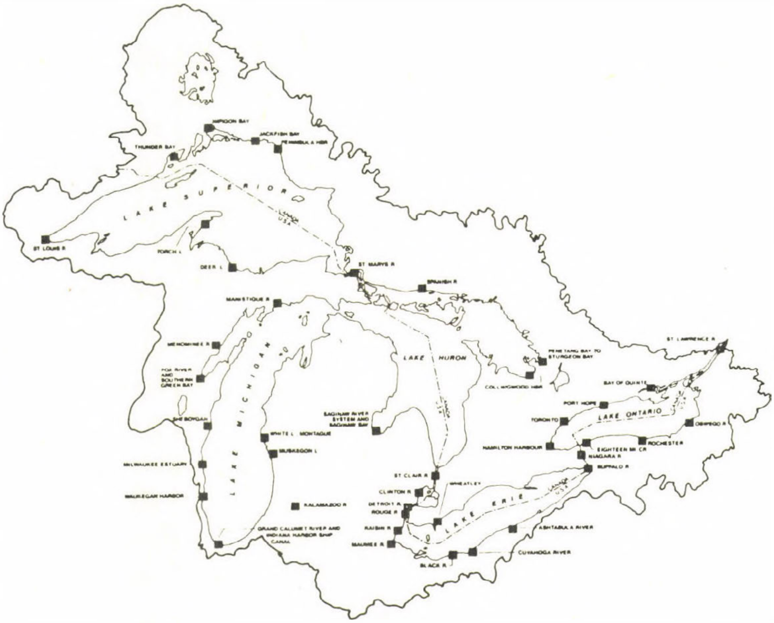


Fig. 7. Areas of Concern in the Great Lakes Basin.

As a result of the 1985 Report of the Water Quality Board, the eight Great Lakes states and the Province of Ontario have committed themselves to developing a remedial action plan (RAP) to restore all beneficial uses in each Area of Concern within their political boundaries. A remedial action plan should identify specific measures necessary to control existing sources of pollution, abate environmental contamination already present and restore beneficial uses.

These programs include, but are not limited to, municipal and industrial wastewater treatment, hazardous waste management, nonpoint source pollution control, groundwater, fisheries and wildlife management, dredging and harbour maintenance, land use planning and recreation. Remedial action plans represent the first systematic and comprehensive effort to restore beneficial uses in the Areas of Concern, and are thus consistent with the ecosystem approach outlined in the 1978 Agreement to protect the waters of the Great Lakes system.

Shifting the focus and emphasis from the open lakes- where improvements are satisfactory - to the Areas of Concern, can be considered a major change in the remedial strategy of the 80's and a challenge for the 90s.

References

Allan, R.J, 1989. Anthropogenic organic chemical pollution of lakes with emphasis on the Laurentian Great Lakes. (This issue).

Allan, R.J. and B.C. Kenney, 1978. Rehabilitation of eutrophic prairie lakes in Canada. Verh. Internat. Verein. Limnol. 20:214-224.

Barica, J., 1981. Lake Erie oxygen depletion controversy. J. Great Lakes Res., 1982.

Barica, J., 1987. Increases of nitrate in the Great Lakes. NWRI Contribution No. 87-77, 12 p.

Bird, P.M. and D.J. Rapport,1986. State of the Environment Report for Canada. DOE, Ottawa, 263 pp.

Brownlee, B.G., D.S. Painter and R.J. Boone, 1984. Identification of taste and odour compounds from western Lake Ontario. Water Poll. Res. J. Canada. Volume 19, No. 1: 111-118.

Bruce, J.P. and P.M. Higgins, 1977. Great Lakes Water Quality Agreement. Progr. Wat. Techn., 9(1): 13-31.

Chapra,S.C., 1977. Total phosphorus model for the Great Lakes. Proc. Am. Soc. Civ. Eng. Div. 103 (EE2): 147-161.

Charlton, M.N., 1980. Hypolimnion oxygen consumption in lakes: discussion of productivity and morphometry aspects. Can. J.Fish. Aquat. Sci. 37(10): 1531-1539.

Dobson, H.F.H., 1981. Trophic conditions and trends in the Laurentian Great Lakes. W.H.O. Water Quality Bulletin, 6(4):146-151, 158 - 160.

Dobson, H.F.H., 1984. Lake Ontario Water Chemistry Atlas. Environment Canada, Inland Waters Directorate, Scientific Series No. 139. 100 p with 94 figures.

Forde, A.V., 1979. A historical study of pollution of Burlington Bay. Reg. Mun. Ham.-Went. Ham. Mun. Lab. Report, Hamilton. 57p.

Forsberg, C., 1979. Die physiologischen Grundlagen der Gewasser-Eutrophierung. Wasser u. Abw. Forsch. 12, 2: 40-45.

Great Lakes Water Quality Agreement of 1972. International Joint Commission of Canada and the United States. U.S. Governm. Printing Office, Washington, D.C. 20402; 69 p.

International Joint Commission (IJC), 1987. 1987 Report on Great Lakes Water Quality. Great Lakes Water Quality Board. Toledo, Ohio. 236 p.

Kwiatkowski, R.E., 1982. Trends in Lake Ontario Surveillance Parameters, 1974-1980. J. Great Lakes Res. 8(4): 648-659.

Kwiatkowski, R.E., 1984. Comparison of 1980 Lake Huron, Georgian Bay - North Channel Surveillance data with historical data. Hydrobiol. 118: 255-266.

Legge, R.F. and D. Dingeldein, 1970. We hung phosphates without a fair trial. Canadian Research and Development, March/April, 1970.

Ontario Ministry of the Environment (MOE), 1988. Severn Sound Remedial Action Plan. Part 1. 99 p.

Munawar, M., and I.F. Munawar, 1986. The seasonality of phytoplankton in the North American Great Lakes, a comparative synthesis. Hydrobiologia 138: 85-115.

Painter, D.S., and G. Kamaitis, 1985. Reduction of Cladophora biomass and tissue phosphorus in Lake Ontario, 1972-1983. NWRI Contribution No. 85-39, Burlington. 13 p.

Rosa, F. and N.M. Burns, 1987. Lake Erie Central Basin Oxygen Depletion Changes from 1929-1980. J. Great Lakes Res. 13(4):684-696.

Robinson, G.W., 1986. Water quality of the Bay of Quinte, Lake Ontario, before and after reductions in phosphorus loading. Can. Spec. Publ. Fish. Aquat. Sci. 86: 50-58.

Stevens, R.J. and M.A. Neilson, 1987. Response of Lake Ontario to reductions in phosphorus load, 1967-82. Can. J. Fish. Aquat. Sci. 44: 2059-2068.

Thomas, R.L. and J.H. Hartig, 1988. Development of Plans to Restore Degraded Areas in the Great Lakes. Environmental Management Vol.12, No.3, pp.327-347.

Vallentyne, J. R., 1970a. Statement presented on behalf of the International Lake Erie and Lake Ontario - St. Lawrence River Water Pollution Boards at the International Joint Commission hearing in Hamilton, Ontario, February 2, 1970.

Vallentyne, J. R., 1970b. Phosphorus and the control of eutrophication. Can. Res. & Develop., May-June, 36-43.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest
Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

ASSESSMENT OF WATERSHED IMPACT AND LAKE ECOLOGICAL STATE FOR PROTECTION AND MANAGEMENT PURPOSES

A. HILLBRICHT-ILKOWSKA

Institute of Ecology, Polish Academy of Sciences
Dziekanów Leśny, 05-092 Łomianki, Poland

INTRODUCTION

Eutrophication still remains the main cause of anthropogenic disturbances on global and regional scale for natural lakes and their environments even after the elimination of the point sources of nutrient input. Particularly, it is the main cause of changes for the numerous small and medium-sized lowland, temperate lakes situated in arable regions and functioning as important tourist centres. This is the case of the Baltic postglacial lakelands to which the Masurian Lakeland in Poland belongs.

The multicriteria evaluation system dealing with trophic impact (in terms of phosphorus loading), role of watershed in supplying and transport of nutrients, natural resistance of lakes against nutrient cumulation and the response of the lake's biota along the trophic continuum was suggested as the operational system for management and protection purposes. The system which is composed of the relevant classes and categories was implemented in 24 lakes situated in the Masurian Landscape Protection Area (Fig. 1) in the Masurian Lakeland (Poland) and the means for different forms of protection of watersheds and lakes were recommended.

WATERSHED IMPACT

The selected physiographic properties and land use in watershed were assessed in the four-point scale, and four classes of watershed impact on lakes were distinguished (Cla WI) (Bajkiewicz-Grabowska 1987). The classification is based on the range of values (see Table 1) of selected properties the role of which in the process of supplying and transporting the nutrients is well documented. The increasing or decreasing values or the changing character of these properties could stimulate or mitigate the surface and underground transport of water and matter to the lake.

Table 1

Cla WI	Mean point value	Watershed impact
1st	≤ 1	Very weak
2nd	1.1-1.4	Weak
3rd	1.5-1.9	Moderate
4th	> 2.0	Strong

Parameter	No. of points 0	1	2	3
1. Ohle's index	<10	10-40	40-150	>150
2. Water balance	-	with outflow	without outflow	flow-through
3. Drainage density ($km.km^{-2}$)	<0.5	0.5-1.0	1.0-1.5	>1.5
4. Average slope (%)	<5	5-10	10-20	>20
5. Share of depressions (%)	>60	45-60	20-45	<20
6. Geological substratum	loamy	sand-loamy	loam-sandy	sandy
7. Land use	forest swamp	forest-arable	arable	arable with urban areas (>10%)

NATURAL RESISTANCE OF LAKE

The trophic impact of the comparable intensity will produce different effects in the lakes of different morphometry and flowthrough regime. To assess these conditions, four categories of lake resistance (Cat LR) were first proposed by Kudelska et al. (1983) and then modified by Bajkiewicz-Grabowska (1987). They are based on the four-point assessment of selected factors (see Table 2) involved in cumulation, bottom release or recirculation of phosphorus load. Cat LR 1st usually represents the deep large lakes and/or high flushing rate; Cat 4th usually shallow, well mixed lakes with small outflow and well-developed contact zone (shoreline).

Table 2

Cat LR	Mean point value	Lake resistance
1st	≦0.8	High
2nd	0.9-1.6	Moderate
3rd	1.7-2.4	Weak
4th	>2.4	Very weak

Parameter	No. of points 0	1	2	3
1. Mean depth (m)	>10	5-10	3-5	<3
2. Ratio: lake volume (10^3 m^3) to shoreline length (m)	>5	3-5	1-3	<1
3. Share of unmixed layer in lake volume (%)	>35	20-35	10-20	<10
4. Ratio: bottom area in epilimnion (m^2) to its volume (m^3)	<0.10	0.10-0.15	0.15-0.30	>0.30
5. Annual water exchange rate	>10	5-10	1-5	<1
6. Schindler's index	<10	10-30	30-100	>100

TROPHIC CLASSIFICATION OF LAKES

The summer concentration of TP in the surface layer was taken (Hillbricht-Ilkowska 1984, Kajak 1983) as the primary discriminant for three classes of lake trophy (Cla LT) but of different ranges for dimictic lakes (i.e. sufficiently deep to be stratified in summer) and for polymictic lakes (i.e. shallow, permanently or frequently mixed in summer down to the bottom). The relation between internal and external loading being usually very high in the former lakes, is the main reason for distinguishing both mictic types of lakes.

Closely related to the increasing range of TP values are other trophic parameters such as water transparency (SD) and chlorophyll concentration (Chl a), phytoplankton biomass (see Table 3) as well as other structural and functional properties of the lake ecosystem such as biomass and share of blue-green algae and nanoplanktonic algae, biomass and composition of zooplankton, especially rotifers, and the interlevel efficiencies in plankton food chain (ratio of predatory to unpredatory biomasses).

Table 3

Cla LT	Concentration in summer, surface layer			
	TP ($\mu g \cdot l^{-1}$)	SD (m)	Chl a ($\mu g \cdot l^{-1}$)	Phytopl. biomass (WW) ($mg \cdot l^{-1}$
Dimictic lakes				
mesotrophic	≦50	≧3	≦10	≦5
moderately eutrophic	≦100	<3	≦30	≦20
strongly eutrophic	>100	≦2	>30	>20
Polymictic lakes				
mesotrophic and/or moderately eutrophic	≦100	≦2	≦30	≦30
strongly eutrophic	≦300	≦2	≦100	≦100
hypertrophic	>300	≦1	>100	>100

LAKE ENDANGERING BY EUTROPHICATION

Vollenweider's concept (1976) of the permissible and dangerous phosphorus load related to the mean depth and residence time of water appeared to be useful for management in most P-limited lakes. Following this concept three categories of lake endangering (Cat END) are suggested (Hillbricht-Ilkowska 1984) based on the relation between the actual annual TP load (i.e. the input with sewage, runoff, precipitation, tributaries) and the permissible and/or dangerous one (both expressed in $g \cdot m^{-2}$ lake area$\cdot y^{-1}$; Table 4).

Table 4

Cat END	TP load relation
1st	the actual load is lower than permissible
2nd	the actual load is equal to or higher than permissible but lower than dangerous
3rd	the actual load is equal to or higher than dangerous

The percentage of TP in point sources (sewage) in actual load is distinguished as being the controllable amount.

MASURIAN LANDSCAPE PROTECTION AREA

An area of about 700 km^2 (Fig. 1), was established in 1977 to protect the land forms (morains, ramparts), forest complexes, marshy and bog habitats, rivers and lakes typical of the postglacial landscape. Among them are: the largest lake in Poland, Lake Śniardwy (110 km^2), biosphere reserve - Lake Łuknajno (with one of the largest population of mute swan in Europe) and other small lakes and marshes forming the reserves for protection of rare plant species and communities and water fowl colonies. The area is protected from the heavy industry, however, is open for arable (including cattle breeding) and forestry activities as well as for touristic exploitation. Because of good natural connection between lakes (River Krutynia)

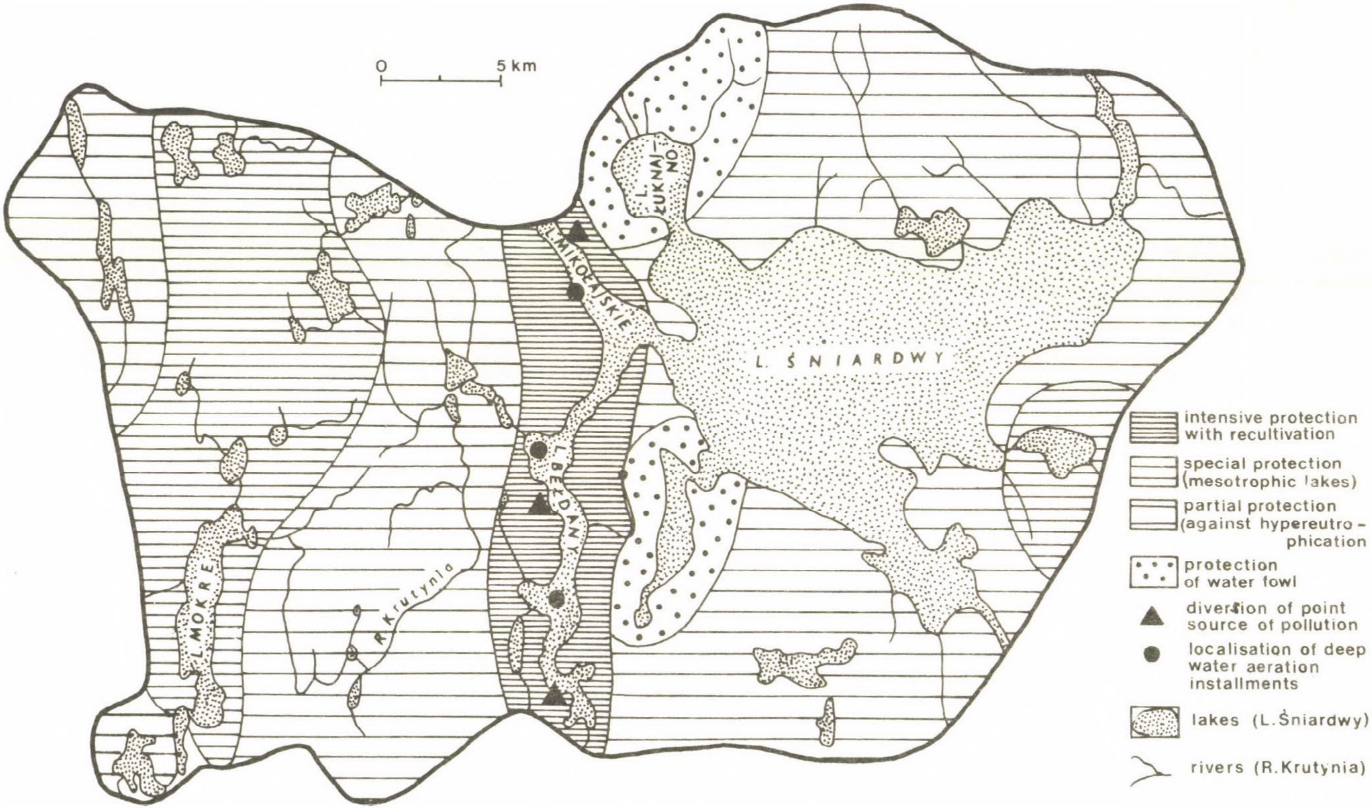


Fig. 1. Masurian landscape protection area: the recommended trends in watershed and lake protection

the area is one of the most popular canoe and sail passages. There are as many as 24 lakes larger than 50 ha (Fig. 1) (and altogether 60 lakes larger than 1 ha) of a mean depth between 0.5 and 30 m, most of them of harmonious meso- and eutrophic character.

The distribution of a number of 24 larger lakes in the Masurian Protection Area according to classes, categories and groups as described above is shown in Table 5.

Table 5*

	1st	2nd	3rd	4th
Class of watershed impact	4	8△	9○●	3▲
Category of lake natural resistance	2▲●	9△○	8	5
Category of lake endangering**	9(0)	2(2)●	13(22)△▲○	
Classes of water purity***	6	3△●	13△○	2
Classes of lake trophy	D-I P-I -11△●	D-II P-II -10	D-III P-III -3▲○	

*The position of four largest (5-110 km^2) lakes: △L. Śniardwy, ▲L. Mikołajskie, ○L. Bełdany and ●L. Mokre is appropriately marked. **In brackets: including the TP load in tributaries. ***According to the classification used by national water control agencies (Kudelska et al. 1983)

For most of the lakes including the largest ones, the 3rd Cat END was assessed which means that the actual TP load is higher (several to over ten times) than the dangerous one. Only one deep lake (Lake Mokre) receives the annual TP load still on the permissible level (2nd Cat END). The TP input from tributaries dominates in the annual load and the sewage contributes significantly (30-70%) only in a few lakes like in Lake Mikołajskie and in Lake Bełdany. It means that for most of the lakes the dangerous level of load is attained only from non-point sources of TP input and from the tributaries, i.e. from uncontrollable sources. In this situation the satisfactory state of water purity (only 2 lakes are out of classification) in most lakes and their mesotrophic or moderately eutrophic status (D-I, P-I, D-II, P-II like Lake Śniardwy) should be the transitory ones. The moderate or weak natural resistance of lakes to watershed impact (mostly Cat LR 3rd and 2nd) may enhance further eutrophication, however, the weak and/or moderate impact of watershed (Cla WI 2nd and 3rd) seems to slow down the actually recorded eutrophication state. However, in the most hazardous conditions are the narrow and deep lakes Mikołajskie and Bełdany with the highest position in all five ranking systems, including the water purity classification.

The following types of protection were recommended:

Intensive protection of watershed and lake recovery (first by deep-water aeration and second by elimination of TP internal loading) and the diversion of principal point sources were suggested for the central part of the area connected with Lakes Mikołajskie and Bełdany. This kind of protection includes also the redistribution of camping places and touristic paths, full protection of small marshy areas (pools, ditches) and forest fragments, control of fertilization and cattle breeding in the arable land, ban for any kind of urbanization.

Special protection of watershed and lakes was suggested for the western part of an area in which several mesotrophic lakes are situated (including the deep L. Mokre) in order to maintain their actual state. Among others, this means: control of fish introduction, full protection of forest in the near lake zones and their exclusion from wood harvesting, de-concentration of the touristic movement as well as ban for its further increase, as also ban for liming and fertilizing the small dystrophic lakes, and finally strict control of agricultural activities.

Partial protection (mainly against hypertrophization) includes some suggestions concerning fishery, forestry and agricultural operations in the watershed and in the lake but with more relaxed suggestions concerning tourism. It was recommended for the eastern part of the area including the watershed of Lake Śniardwy.

Lake protection related to waterfowl protection is suggested for two shallow lakes (including L. Łuknajno - biosphere reserve), which means a ban for any touristic movement in the vicinity of the lake as well as for any intervention into the trophic state of the lake and the rate of its overgrowing.

REFERENCES

1. Bajkiewicz-Grabowska, E. 1987. Natural degradation ability of lakes and the role of drainage area in this process (in Polish, with English summary). Ekol. Pol. 33, 280-289.
2. Hillbricht-Ilkowska, A. 1984. The indices and parameters useful in the evaluation of water quality and the ecological state of temperate, lowland lakes connected with their eutrophication. Proc. Int. Conf. on "Conservation and Management of World Lake Environment". Otsu, Shiga, Japan, 24-30 August 1984, pp. 55-69.
3. Kajak, Z. (ed.) 1983. Ecological characteristics of lakes in North-Eastern Poland versus their trophic gradient. Ekol. Pol. 31, 239-530.
4. Kudelska, D., Cydzik, D., Soszka, A. 1983. Lake water quality classification system (in Polish). Publ. Int. Environment Management, Warsaw, 44 pp.
5. Vollenweider, R. 1976. Advances in defining critical level for phosphorus in lake eutrophication. Mem. Inst. Ital. Idrobiol. 33, 53-83.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

ANTHROPOGENIC LOAD AND EUTROPHICATION OF THE KUIBYSHEV RESERVOIR

S.M. KONOVALOV, L.A. VYKHRISTYUK, V.N. PAUTOVA, V.I. NOMOKONOVA, M.M. VYKHRISTYUK

Institute of Ecology of the Volga River Basin, Academy of Sciences, 445003, Togliatti, USSR

The Kuibyshev Reservoir on the Volga River is one of the largest in the world, in terms of the water surface area (5,900 km^2). Its mean depth is 9 m (the maximum being up to 40 m), its water exchange rate amounts to four times a year. Water masses are turbid, the transparency rate seldom exceeding 2 m (Fig. 1).

The annual nutrient load on the reservoir is 7 g/m^2 and 61 g/m^2 of total P and total N, respectively (Fig. 2). The seasonal dynamics shows that their maximum concentrations occur during the spring floods (Fig. 3). Unlike N_{total} which concentration in the reservoir tends to reduce by the fall, the level of P_{total} begins to grow again after the summer decrease. The N:P ratio during the period of open waters changes accordingly; in other words, the algal growth can be phosphorus- and nitrogen-limited.

Presently, the average concentration of P_{total} in the reservoir amounts to 0.103 mg/l, and that of N_{total} is 1.20 mg/l. These values are ca. 30% higher than the ones recorded throughout the 60s-70s. The construction of Nizhnekamsk (1979) and the Cheboksary (1980) power stations on the Kama and Volga Rivers, respectively, caused a rapid increase of nutrient concentrations in the Kuibyshev Reservoir during the 80s.

All this has accordingly caused alterations in the phytoplankton composition (see "Ecology of Phytoplankton of the Kuibyshev Reservoir", 1989). While the dominance of Melosira, Stephanodiscus, Cyclotella in the early spring phytoplankton in different years and ratios was maintained as before, intensive

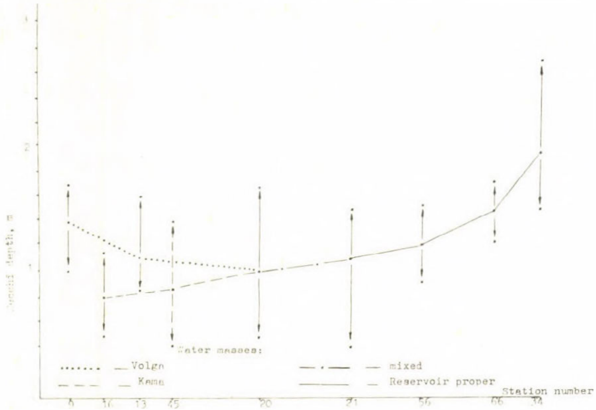


Fig. 1. Variations of water transparency along the Reservoir axis.

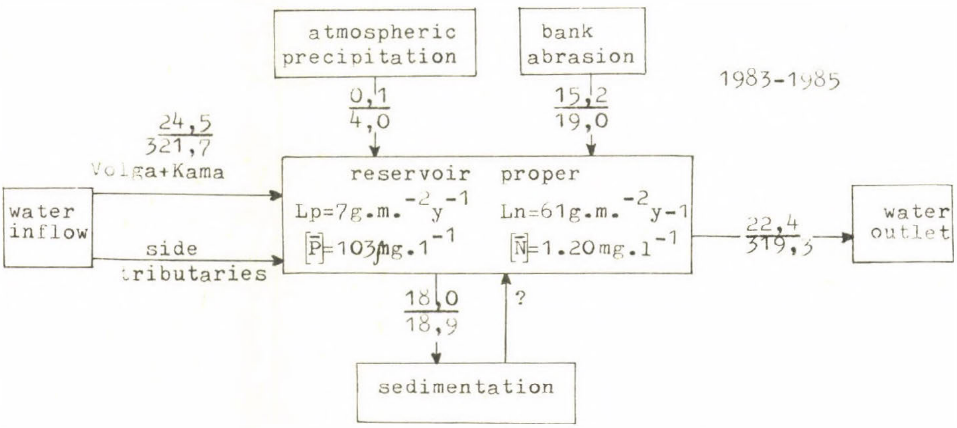


Fig. 2. Elements of phosphorus (numerator) and nitrogen (denominator) balance in the Kuibyshev Reservoir (thousand tons/yr).

growth of Pyrrophyta in the presence of Bacyllariophyta and Chlorophyta in late spring became regular occurrences, which is typical of eutrophication-impacted water bodies. During the 80s at the background of abundant Bacyllariophyta and Chlorophyta and unconditional dominance of Aphenizomenon flosaquae and Microcystis aeruginosa in the reservoir proper, there were increasing instances of Anabaena, Lyngbya, Coelosphaerium as well as of smaller Stephanodiscus, all being indicators of a

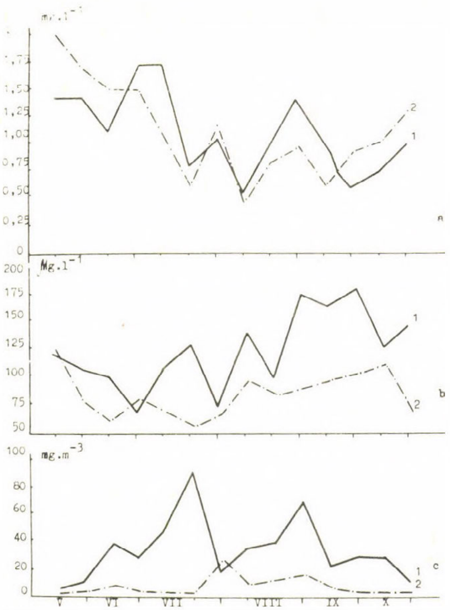


Fig. 3. Seasonal dynamics of N_{total} (a), P_{total} (b) and Chl "a" (c) concentrations in the Reservoir water masses: 1 - The Cheremshan Bay; 2 - The transit part of the Reservoir.

water-body eutrophication. During the fall one can infrequently observe the water "bloom" caused by Melosira and Stephanodiscus spp.

The increasing nutrient concentrations could not help impacting the biomass of planktonic algae as well, but its assessment was effected only once a month throughout the sparse net of surveillance stations. Due to the "patchiness" in the phytoplanktonic algae distribution, typical of large eutrophic water bodies, the above observations failed to reveal any pronounced trend in phytoplanktonic biomass increase from the 70s towards the 80s. Starting from 1985 the very procedure of observations on the state of plankton underwent certain changes: its assessment involves nowadays not only conventional count-and-volume method, but chlorophyll "a" concentrations as well. Absolute maximum of chlorophyll "a" concentrations amounted to 180 g/m^3 (1985) which is approximately equal to 90 kg/m^3 of

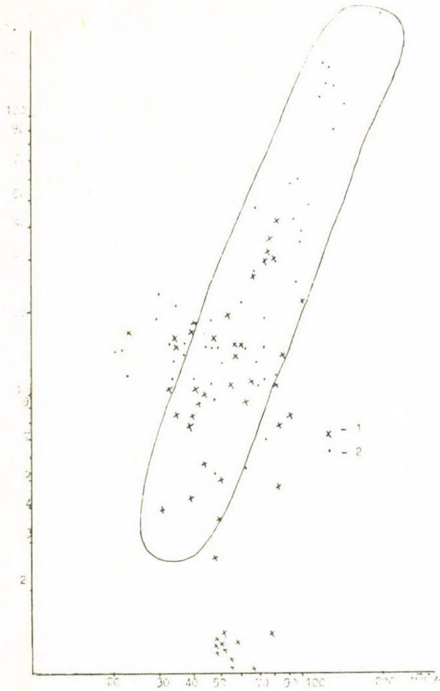


Fig. 4. Chlorophyll "a" to total phosphorus relationship in the Reservoir water masses (1) and Cheremshan Bay (2), June-September, 1985, 1987.

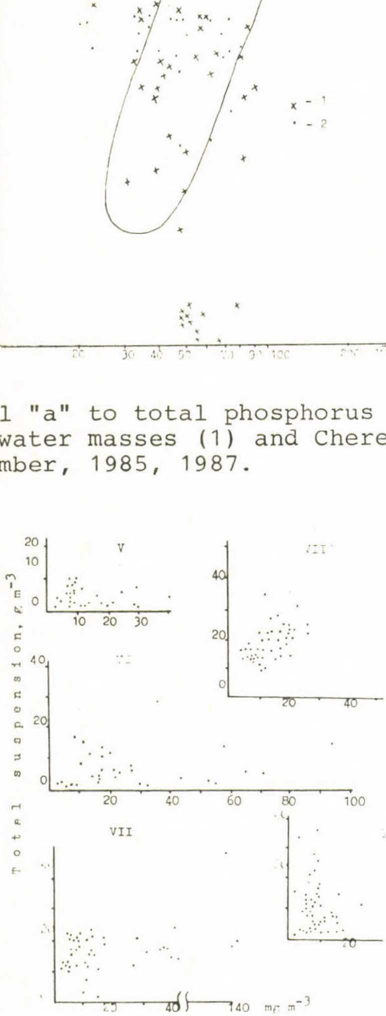


Fig. 5. Relationship between suspension and Chl "a", 1987.

the biomass. This is the highest value throughout the whole period of the phytoplankton studies in the Kuibyshev Reservoir. The estimated equation of chlorophyll "a" to P_{total} relationship (Fig. 4) is as follows:

$$\text{Chl."a"} = -1.5 + 1.28\, P_{total} \qquad (r = 0.70).$$

The same relationship as referred to N_{total} is less pronounced.

When studying the mechanisms of self-purification in the Kuibyshev Reservoir, particular attention has been given to sedimentation processes and accumulation of nutrient compounds in the bottom sediments. Maximum amount of total particulate matter is typical of flood waters. In the process of the reservoir water level raising and current velocity decreasing, the amount of particulate matter would reduce to less than 10 g/m^3 due to its settling down. Development of Cyanophyta during the summer time gives rise to a further increase in suspension concentrations (Fig. 5).

Judging by the depth of silt deposits a greater portion of particulate matter settles down in the lake-shaped pools (upstream of the Preplotinny [predam] Pool). Each square meter of its bottom sediment area is added by 3 kg of total particulate matter, 54 g O of liable organic matter (as per BOU_5 - biochemical oxygen uptake) and 0.3 g of chlorophyll "a". This was established using sedimentation traps.

Average rate of deposit accumulation determining the sediment uptake capacity comprises 0.4 cm/yr. Deep-water sections are characterized by a greater accumulation rate, so the depth of deposits there reaches 100 cm, being no more than 20-40 cm in the flood-plain. Annual nutrient inflow into the bottom sediments includes 19,000 t and 40,000 t of P_{total} and N_{total}, respectively, i.e. 46% and 11% of their total inputs into the reservoir.

P_{total} and N_{total} concentrations in the bottom sediments of the lower section of the reservoir are presented in Fig. 6. It is obvious that their amounts in the sites of Togliatti wastewater discharge abruptly increases. The sediment pollution area covers ca. 15 km^2. As to the water masses themselves, the

Fig. 6. P_{total} and N_{total} concentrations in the upper layer of bottom sediments in the Reservoir.

impact of pH pollutions and electroconductivity is distinctly traced over a lesser zone in the vicinity of the discharge sites.

Nutrient release from the bottom sediments has been determined for phosphorus only. In summer its daily amount reaches 2 mg/m^2, i.e. the threshold beyond which there begins the diffusion of nutrients back into adjacent water columns.

The Volga and Kama waters are responsible for the major portion of total P and N inputs; about 30% of the total nutrient load comes from the reservoir drainage area.

It is obvious that any input management within the frames of the Kuibyshev Reservoir alone would be insufficient. Nutrient inputs become involved into the general cycling of substances in the aquatic ecosystem, and therefore can be rendered as transit. They ultimately reach the Caspian Sea causing marked alterations in its biota. In terms of strategy, measures should be taken, first and foremost, on the reduction of nutrient load coming from the upper reaches of the Volga and Kama Rivers.

REFERENCE

Ecology of Phytoplankton of Kuibyshev Reservoir. Nauka, Leningrad 1989, p. 207.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

EUTROPHICATION OF SMALL LAKES IN LATVIA

R. LIEPA, P. CIMDIŅŠ, V. RODINOV

Institute of Biology of the Latvian SSR Academy of Sciences, 229021 Salaspils, 3 Miera str., Latvian SSR, USSR

Nowadays measures taken against eutrophication of lakes to conserve their trophic state should be based on the comprehensive study of the lakes and the hydrographic net of their drainage area. Thirty-seven lakes in various parts of Latvia were investigated. The lakes differ in morphologic and morphometric indices, and in the degree of their trophic level and pollution. However, all of them are more or less subjected to the impact of economic activities in the drainage area or in the lakes themselves, and being small ones, they are mainly used for recreation purposes.

On soil erosion or as a result of the irrational use of fertilizers, agriculture loses annually a considerable amount of mineral and organic substances. The methods applied for soil protection against erosion are by far not as effective as the intensity of agricultural activities.

All kinds of soil-formation processes occur in Latvia, but the largest area (about 60%) is covered by soddy podzolic soils. In general the soils in Latvia are not rich in nutrients and they require a great amount of mineral fertilizers.

Latvia is situated in the zone of excess moistening, therefore most of the soils are periodically or constantly overdrenched. The present area belongs to the region of widely used artificial drainage which is connected with agricultural development: the majority of the drainage area of lakes and rivers are under cultivation. In order to maintain an optimal regime for plant nutrition under such conditions, great importance is attached to the control and compensation of elements washed out by precipitation or consumed by plants. Studies on the leaching of elements from the soil have revealed that the cultivation layer mostly loses potassium, calcium, magnesium and phosphorus at monthly precipitation of 45-78 mm. It should also be considered that in the European part of the USSR, including Latvia, the amount of monthly precipitation reaches 100 mm and even more in the different periods. This produces considerable losses of nitrogen, potassium, magnesium, copper, zinc, manganese, molybdenum, and boron even in the initial stage of precipitation.

Loss of nitrogen and phosphorus compounds, either by precipitation or by drainage, results in the eutrophication of water bodies. Under the impact of drainage the level of nitrate nitrogen in the surface waters sometimes increases to above 200% and that of mineral phosphorus to above 500% as compared to the level prior to fertilization. Due to the phosphorus fertilizers spread over the snow in winter and carried away by the runoff, the level of phosphorus in the lake water increases more than hundredfold.

Biogenous substances entering the water in various ways cause structural as well as functional changes in communities. There is a growing number of sources of pollution due to economic activities which can be traced even in the distant water bodies within woods and swamps intact so far.

The basic research on lake eutrophication has been made in two interrelated directions, by studying (i) physicochemical and biological properties of drainage waters and runoff from agricultural lands, and (ii) the processes and consequences of lake eutrophication by drainage waters to develop methods for preventing the eutrophication of Latvian lakes. In the first case investigation was carried out in the field, while in the second, in situ and experimentally. Detailed studies on the impact of drainage waters and agricultural runoff have been made in two lakes of different type. Kiruma, a highly eutrophic lake, is situated in the drainage area of the River Salaca, with drainage waters of the experimental melioration area of 63 ha running into it. There are 25 types of experimental drains installed at various depths and distances.

The oligotrophic Lake Vaidava belongs to the area of the River Gauja. The basic source of biogenous elements is the surface runoff with fertilizers during spring and autumn, and it is mainly determined by soil erosion.

The ecologic condition of lakes is estimated by the aggregation of living organisms in the water, i.e. zooplankton, protozoans and benthos. Their structural and functional changes are related to the steadily increasing concentration of biogenous elements drained from the drainage area of lakes.

In order to establish the factors causing and limiting eutrophication, experimental studies were carried out in two small lakes, Kiruma and Vaidava, to estimate the ecologically based norms of biogenous elements. Communities of water organisms were investigated in isolated amounts of water (in plastic bags of 100 l capacity) after adding mineral fertilizers (N - 12%, P_2O_5 - 12%, K - 12%) in various doses to reach the concentrations of 1, 2 or 6 mg/l. All links of the trophic chain were studied starting with the autotrophic and ending with the last one of the given microecosystem, i.e. the zooplankton.

To avoid eutrophication recommendations have been worked out for the economic activities in the area of small lakes based on long-term investigations and taking into consideration the regional peculiarities of Latvian lakes.

The limnological character of the lake, especially of a small one, is greatly determined by processes occurring in the drainage area. According to their morphology, the investigated lakes are very different, but with some exceptions, almost all of them have the same typical banks, i.e. sloping banks alternating with steep ones. The slopes stretch from a few to about 900 m, and their steepness exceeds 40° in some places. The slopes of some lakes are covered by shrubs or woods, while others are under cultivation or are pastures. Separate farms, compartment houses, settlements or cattle-breeding farms are to be found on them and camping grounds in the recreation zones. At some lakes the erosion substrate covers terraces, sometimes it is boggy.

All the allochthonous substances first enter the littoral zone of the lake. Therefore, in oligotrophic water basins a zone of increased productivity is formed along the shores with a growing number of Infusoria having a short life-cycle. According to the number of species in oligotrophic lakes, Infusoria decrease in the direction from the littoral to the pelagic zone and from the epilimnion to the hypolimnion. In highly eutrophic lakes the total number of plankton Infusoria is five times higher. There is vertical density of organisms here, but in oligotrophic lakes a maximal amount of Infusoria is accumulated in the euphotic layer. The majority of zooplanktons are unfit for life in highly trophic waters. Under such extreme conditions the less susceptible species of zooplankton survive (including Infusoria), which, not encountering competition for food, reach high numbers.

The variety of species for benthos Infusoria in oligotrophic lakes is three times higher than in the highly eutrophic lakes, their density being, however, 11 times lower. The amount of profundal fauna in highly eutrophic lakes is determined by the degree of water mixing, i.e. the gas regime. In shallow lakes with a relatively large water surface, the water mass is almost always well mixed up and there is no long-term stratification. However, under certain hydrometeorological conditions, short-term stratification may occur and sometimes near the bottom almost all the oxygen is consumed by the degrading organic substances. Under experimental conditions, none of the Infusoria chose a layer without oxygen. In nature, however, there are Infusoria capable of living with a minimal amount of oxygen in the water and some of them even under anaerobic conditions (sapropelebiotic Infusoria). In oligotrophic water bodies these forms occur sporadically in the littoral zone where decomposition of organic substances starts. The littoral zone acts as a filter for biogenous matter.

In contrast to water macroorganisms, which respond to changes in the environment only after a rather long time, Infusoria respond very quickly. Therefore, their qualitative and particularly quantitative changes can serve as indicators of both the pollution and eutrophication of water bodies. This refers first of all, to the protozoobenthos, since it is known that the 'precipitation' of organic substances is concentrated near the bottom layer and at the bottom. In a highly eutrophic lake, cenoses of protozoobenthos form similar to those of the polluted rivers in Latvia, but the cenoses of oligotrophic lake littorals resemble those of a moderate pollution. Under saprobic conditions, there is an intensive destruction of organic substances while under trophic ones a high production of these substances, as also revealed by the data of zooplankton research.

In a eutrophic-polytrophic lake the presence of zooplankton is indicative of a low trophic state in the littoral area and a high one in the pelagic area. It can be explained by the overgrowing of sloughs due to the development of rotifers, i.e. indicators of xenooligosaprobic conditions. However, this is periodical and is only connected with the coincidence of cycles of zooplankton and vegetation development under atmospheric aeration. Finally, due also to a possible bacteriostatic effect of the apical plant shoots, favourable oxygen and food conditions are provided for a number of oligosaprobic species. During the autumn-winter decay this zone corresponds to a polysaprobic state, while the cycle of oxygen consumption in the destruction of vegetation coincides with the lack of atmospheric aeration under the ice. Thus, the joint impact of factors such as the production and consumption of oxygen, may largely predetermine the state of the lake.

A difference in food web functions is considered to be the most typical characteristic of polysaprobic and eutrophic lakes. In a eutrophic water body, nutrients of allogeneic origin are well absorbed in the food web and they maintain heterotrophic production on a high level. In a polytrophic water body allochthonous nutrients are only partially absorbed in the food web, their excess amount distinctly stimulates the autotrophic production and limits the production of heterotrophic substances.

Solving the problem of de-eutrophication by biological means, particularly by liquidating the excess production with the help of a more complicated trophic chain (fish acclimatization and maintenance of their natural food basis), the causes of disturbance in the normal functions of the food web should be considered more in detail. The toxicity of blue-green algae is regarded to be the basic cause. Microcystis aeruginosa in Lake Kiruma reaches its maximum development in August, but the development of zooplankton is reduced at M. aeruginosa of 2-4 mg/l. Not all blue-green algae are fit for filtrate food, only M. aeruginosa is of a toxic effect. Brachionus calyciflorus is an exception, which not only feeds on these algae but also maintains the density of its population on a high level. Rotifers of this type are also characteristic of Lake Kiruma and, obviously, for promoting their consumption of blue-green algae, such indicators as pH and oxygen level in the water should be sufficiently high in the water body. It will result in the development of filtrates and a reduced trophic level in the lake.

By increasing the concentration of biogenous elements in the experimental isolated volumes, not only the composition and proportionality of dominating zooplankton groups (including Infusoria) undergo changes. In the dominant group there is no alteration due to the direct impact of growing concentrations of mineral fertilizers since no morphologic changes have been observed. The total amount of Infusoria is directly proportional to the growth of biogenous substances in the water. However, a large number of Infusoria at a maximum concentration of biogenous elements is due only to one species, Coleps hirtus. Reproduction of Infusoria in all the experimental (including also the control) volumes is lower than in the lake. Obviously, it can be partially related to the limited space lacking an exchange of water mass and disturbing the natural relationship between water surface and substrate. There are several drawbacks in the microsystem, being a model of the lake, i.e. abundant development of periphyton the biomass of which exceeds that of other components, accumulation of metabilic products of all water organisms, increased consumption of Infusoria not only by zooplankton but also by chironomid larvae occurring in extreme amounts. A negative impact of the blue-green algae can also be mentioned resulting in the formation of high-molecular compounds in the environment inhibiting the life functions of other water organisms. Interacting with physiologically active substances, they are of great importance in determining the structure and quantitative composition of biocenosis. In the experimental volumes, due to a rich supply of biogenous elements, an abundant development of tiny Infusoria begins with a very brief life-cycle, and the trophic structure of the whole microplankton community is simplified.

Methods for de-eutrophication are selected according to the regional conditions, i.e., morphology of the water body and its drainage area, level of eutrophication and technical possibilities. The biological method of de-eutrophication makes use of the food web for eliminating excess biogenous material, though it partially solves the problem of producing food proteins (fish). It is efficient only at the initial stages of eutrophication and at the phosphorus level 0.003-0.015 mg PO_4^{3-}/l.

In order to plan the protection and rational use of lakes in Latvia, the disordered development of water bodies should be prevented, which, under an increased anthropogenic impact, occurs very quickly and results in a drastic deterioration of the ecological condition of lakes. With planned management of the development of lakes and rivers a regulation of the circulation of biogenous elements is supposed in the drainage areas and a regulated inflow of biogenous elements in the water bodies. To manage the development of various types of lakes situated in agricultural areas, and recreation zones and the protection of nature, practical recommendations are to be worked out:

1. To prevent the spread of fertilizers on frozen soil, over the snow and close to the banks of rivers, lakes and canals joining them in the drainage areas of oligotrophic lakes and lakes in the recreation and nature protection zones.

2. To prohibit cattle-breeding (large plants) and construction of buildings for mass tourism on drainage areas of importance in recreation and nature protection, if the specific drainage slope (drainage area/lake surface area) is more than 15° or the surrounding fields slope at more than 8°. Fecal fer-

tilizers and runoff from open drainage should not be allowed to get into lakes.

3. Intensive economic activities should be allowed on drainage areas of meso- and eutrophic lakes and of lakes of production importance keeping a buffer zone along the shores at a width of 20-200 m and ensuring that pesticides and herbicides used in agriculture should not enter the water body. Only grass may be cut in the buffer zone. The width of buffer zones should be determined according to the biofunctions of lakes and the steepness of the surrounding area. At recreation and nature protection lakes the buffer zone should be as wide as 200 m, while at production ląkes it should exceed 20 m. In case of a possible overgrowing of lakes, with the inflow of biogenic material exceeding 0.5 kg P/ha, and a lake depth below 7 m, the impact of runoff should be compensated by measures of technical and biological melioration: (i) Technical melioration, i.e. draining the runoff from agricultural lands by a system of by-pass canals and accumulation ponds, should be worked out. The accumulated water can be used for overhead irrigation and the sediments for field fertilization; (ii) Biomelioration, i.e. development of fish-breeding in the accumulation ponds by acclimatizing herbivorous fish, and in lakes by restoring valuable local fish species and (iii) to restore the eutrophic lakes, selective draining should be used, i.e. water drainage from deep layers (above 7 m) without touching the water from the upper layer (1-0 m).

4. Polytrophic lakes of industrial importance, the restoration of which is uneconomical can be used for accumulation and the draining water for technological purposes in agriculture. If necessary such water bodies can be partially purified by the application of surface and deep aeration and by removing the sludge.

Planned transformation of lakes (technical and biological melioration, restoration) should only be done according to ecologically co-ordinated projects developed for separate water bodies. The limited economic activity mentioned in the recommendations in the drainage areas of recreation and nature protection lakes should also apply to the drainage area of rivers flowing into these lakes.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

THE EFFECT OF REDUCED SEWAGE INPUT ON THE TROPHIC DEVELOPMENT OF AN OLIGOTROPHIC LAKE (ATTERSEE, AUSTRIA)

O. MOOG

Department of Hydrobiology, Fisheries Management and Aquaculture, University of Agriculture
Feistmantelstr. 4, A-1180 Vienna, Austria

Abstract: The trophic state of oligotrophic Attersee changed to ultra-oligotrophy after the reduction of the nutrient input to about one third of Vollenweider's critical load.

Attersee, located in central Austria 70 km east of the city of Salzburg (Fig. 1), is the last link in a system of four naturally coupled lakes, namely Fuschlsee, Irrsee, Mondsee and Attersee, contributing to the River Traun watershed. The catchment area is of mountainous character; the lake is 171 m deep, stratified and dimictic. Hydrologic and morphometric data of Attersee are shown in Table 1.

FIG. 1: MAP OF EUROPE WITH AUSTRIA AND LOCATION OF ATTERSEE

TABLE 1:
ATTERSEE:
HYDROLOGIC AND MORPHOMETRIC DATA

ALTITUDE A.S.L	(m)	469.2
TOTAL CATCHMENT AREA	(km)	463.5
DIRECT CATCHMENT AREA	(km)	164.2
LAKE SURFACE AREA	(km)	45.9
LAKE VOLUME	($10^6 m^3$)	3943.0
SHORE LENGTH	(km)	53.0
MAXIMUM DEPTH	(m)	171.0
MEAN DEPTH	(m)	84.0
MEAN OUTFLOW	(m^3/s)	17.5
MEAN RETENTION TIME	(years)	7.2

The region is part of the Salzkammergut lake district. The charming countryside and many possibilities for water-bound recreation characterize recreation characterize this traditional vacation region. The location of the lake near Austria's main highway (A1) facilitates one-day or weekend-trips from two federal state capitals (Salzburg, Linz) and a larger town (Wels).

Day excursions together with an increasing number of holiday tourists have led to a six-fold increase in tourism between 1950 and 1980 (Fig. 2). Finally, by the seventies, the summer population densities were 7 times higher per km^2 than during the off-season. The number of lake area residents also increased steadily.

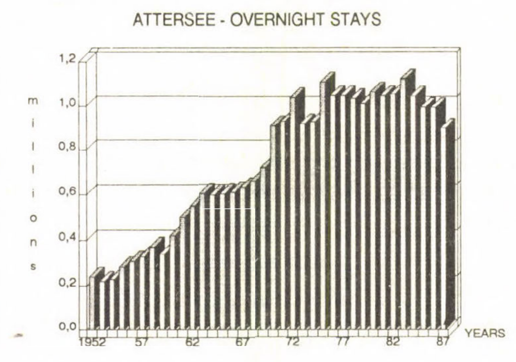


FIG. 2: ATTERSEE - OVERNIGHT STAYS

Increased nutrient inputs due to increased tourism and weekend recreation, as well as a change in the amount and composition of domestic sewage (polyphosphates) led to a cultural eutrophication and deterioration of the water quality of most Salzkammergut lakes. The first changes in water quality were observed in 1961 for Mondsee and 1966 for Fuschlsee. The first algal blooms with hypolimnetic oxygen depletion occurred in 1968 in Mondsee (DANECKER 1969) and 1971 in Fuschlsee (HASLAUER 1979). Attersee, the deepest of the four lakes and situated at the end of the lake chain, still remained oligotrophic in those days but, according to MÜLLER (1976) showed signs of increasing eutrophication.

Alarmed by algal blooms and fish-kills, which negatively influenced tourism in nearby lakes, "lake purification associations" were founded in 1965 (Attersee-region), 1968 (Mondsee-region) and 1973 (Fuschlsee-region). As early as 1964, first investigations on the mode of sewage diversion and treatment were carried out for the Mondsee-Irrsee-region (FLÖGL & KOLMHOFER 1976), 1965 for the Attersee-region (FLÖGL 1976) and 1973 for the Fuschlsee-region (HASLAUER, MOOG & PUM 1984).

The early reactions of politicians to keep the water of Attersee clean was based not only on touristic considerations and their economical consequences but also on the considerable importance of Attersee as a large reservoir of clean potable water.

The chosen sanitary solution was the diversion of sewage by ring canalization (pipes in soil as well as at the lake bottom), with central treatment outside the catchment area in a treatment plant. The sewage treatment plant is designed for 60.000 inhabitant equivalents and its capacity can be expanded to 120.000. It went into operation in 1976. Between 1978 and 1981 the majority of the towns and villages around the lake were connected. As of 1985 even little villages and individual homes have been connected to the diversion system (Table 2).

TABLE 2:
DEVELOPEMENT OF INHABITANTS AND OVERNIGHTSTAYS (PERCENTAGE NUMBER) CONNECTED TO SEWAGE DIVERSION

	INHABITANTS	SUMMER RESIDENCES	OVERNIGHT STAYS
1975	0.0	0.0	0.0
1976	8.5	8.5	8.5
1981	54.9	60.0	53.6
1985	68.2	70.0	68.7

Of the nutrients, only phosphorus must be considered here: with an N:P ratio of more than 130 to 1 (atomic-ratio), only a P-limitation is to be assumed. Within an eight year observation period a distinct decrease in the phosphorus load is evident (Table 3).

TABLE 3:
PHOSPHORUS BUDGET OF LAKE ATTERSEE 1978 - 1985
(kg TOTAL - PHOSPHORUS PER YEAR)

	1978	1979	1980	1981	1982	1983	1984	1985
TRIBUTARIES SUM OF P - INPUT BY DIRECT CATCHMENT	6775	8120	7980	6970	4215	3315	3295	4890
MAIN INFLOW (MONDSEEACHE)	7750	11020	7800	8690	5010	3300	3560	4980
ATMOSPHERIC INPUT	1900	2540	2320	1470	1580	1495	1840	1490
GROSS INPUT	16420	21680	18100	17130	10850	8110	8695	11360
EXPORT (RIVER AGER)	2980	4780	4500	4850	3530	3220	2250	3105
NET - LOAD	13440	16900	13600	12280	7275	4890	6445	8255

From 1978 to 1981 the mean P-load was 14 tons per year; since 1982, P-inputs did not exceed 9 tons per year. Presently the P-input constitutes only 45-60% of the previous load (yearly aver 1982-1985: 6700 kg). The raw data and methods used are presented in MOOG (1982, 1987).

The main nutrient source for Attersee is the inflowing outlet of Mondsee (Mondseeache), which drains a catchment area of 253.4 km^2. The P-load of Mondseeache represents 45.6% of the gross-P-load and 62.8 % of the net P-load. Due to the sanitation of the Mondsee catchment area (ring canalization), the water quality of this lake improves steadily. The yearly volume weighted averages of Mondsee lake water total-P concentrations indicate a 75 % reduction between 1978 and 1986 (Fig. 3). Simultaneously a decrease in the total-P input of Mondseeache into Attersee can be observed (Table 3). A detailed nutrient budget of the Mondsee area is to be published by JAGSCH, MOOG & DOKULIL (in prep).

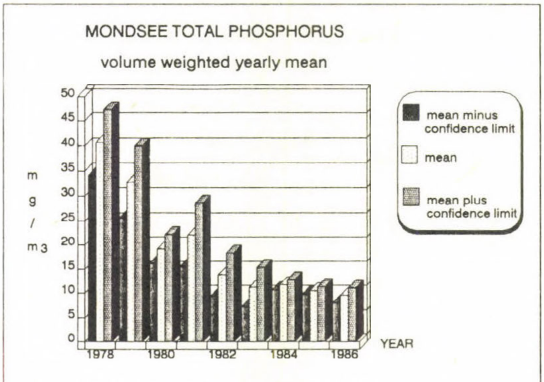


FIG. 3: MONDSEE - TOTAL PHOSPHORUS CONCENTRATION

In relation to the total-P-loads from the direct catchment area (the catchment area which directly drains into Attersee, without the catchment areas drained by Mondseeache), the calculated domestic inputs amount to 64 % in 1981 and only 55 % in 1985. Compared to the number of connections in 1981, an additional 20% of the permanent residents, 22 % of the overnight stays, and 16 % of the summer residences were connected by 1985. Accordingly, the input measurements (Table 3) indicate a 30% reduction of the P-load from the direct catchment area between 1981 and 1985.

The phosphorus import from the direct catchment area of Attersee also depends on the amount of farmland. Agricultural land use contributes 18-20% to the total load. In addition, a high correlation between cultivated land and the phosphorus exports of single drainage basins could be detected (MOOG 1984),(Fig. 4).

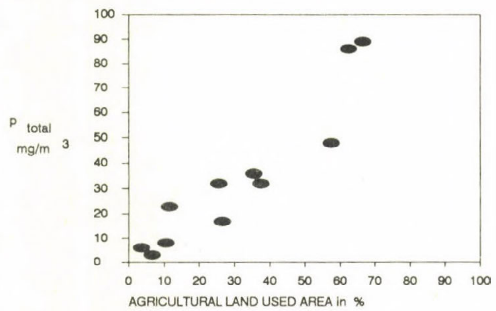


FIG. 4: AGRICULTURAL LAND USED AREA IN % VERSUS TOTAL PHOSPHORUS CONCENTRATION OF TRIBUTARIES:
$y = 1.040x + 2.727$, $R = 0.91$, $N = 11$

Atmospheric input represents another important P-source. The nutrient input by dry and wet depositions onto the lake surface contributes 11% to the gross P-load before and 17% after sewage removal (MOOG, in prep.). Compared to the P-load of the direct catchment area the atmospheric input contributed 17.6% in 1981 and 24.9% in 1985. This is a considerable fraction when one takes into account that the area is not industrialized.

Figure 5 shows the critical load according to VOLLENWEIDER (1975) and the P-load in mg total-P per m^2 lake surface and year. On the average, 70 % of the critical load was attained before sanitation 1978-1981, 36 % (ave. 1982-1985) after sanitation.

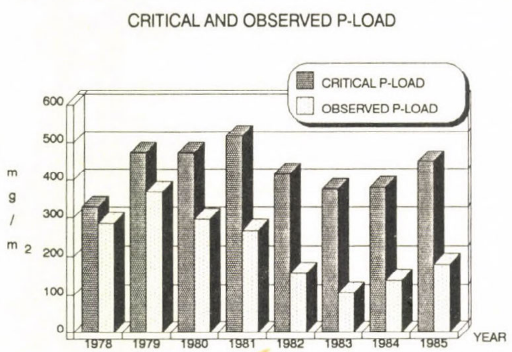


FIG. 5: ATTERSEE - CRITICAL AND OBSERVED P - LOAD

The P-load of Attersee has been estimated both by measurements and by calculations. The calculations are based on export coefficients (Table 4). The coefficients were taken from GÄCHTER & FURRER (1972), HAMM 1976 a,b and RECKHOW et al. (1980), for detailed information see MOOG & SCHINDLBAUER (1982).

The domestic phosphorus load of permanent residents, overnight visitors and summer residences (Table 5) has been added to the exports of the variously utilized areas (Table 6). These calculated P-exports are compared with the results of the observed phosphorus input through tributaries and diffuse sources (Tables 3 and 7).

TABLE 4:
PHOSPHORUS EXPORT COEFFICIENTS IN kg P total PER km^2 AND YEAR

HUMAN OUTPOUT PER DAY	3g P total PER-CAPITA
FORESTS	1 kg
PASTURE	30 kg
ARABLE LAND (ROW CROPS)	70 kg
URBAN RUNOFF	100 kg

TABLE 5:
DOMESTIC PHOSPHORUS LOAD (CALCULATED BY ASSUMPTION OF 3g P total PER CAPITA AND DAY)

	1981	1985
INHABITANTS	4300	3220
OVERNIGHT STAYS	1090	710
SUMMER RESIDENCES	180	140

TABLE 6:
ESTIMATION OF PHOSPHORUS EXPORT BY EXPORT COEFFICIENTS (kg/km^2.year)

	1979		1983	
	AREA	kg/year	AREA	kg/year
PASTURE	25.46	764	24.70	741
ROW CROPS	6.47	453	5.95	417
FOREST	129.05	129	129.31	129
RESIDENTIAL AREA	2.41	241	341.00	205*
TOTAL		1587		1492

*P - export minus share of purified or diverted runoff (60%)

TABLE 7:
COMPARISON OF CALCULATED AND MEASURED LOADS (P total IN kg PER YEAR)

YEAR	CALCULATED LOAD	MEASURED LOAD
1981	7160	6970
1985	5560	4890

The calculated and observed P-loads coincide well. The difference between these two methods of loading estimates is 3 % in 1981 and 13 % in 1985. This narrow span between observation and calculation is within the same order of magnitude as shown in a similar study of Fuschlsee where, within a five year investigation period, a difference ranging from -11 and +14 % between predicted and observed load was reported (HASLAUER, MOOG & PUM 1984).

The limnological development of Attersee from 1978 to 1986 under reduced phosphorus loading conditions is presented in Figs. 6- 8. Secchi depth, chlorophyll A-concentration and total-P-concentration are the parameters used for estimating trophic conditions.

Analogous to the reduced net P-loading, the biomass of the planktonic algae, expressed as chlorophyll A-concentration, decreases steadily (Fig. 6) from 1.35 mg/m^3 (ave. 1978-1981) to 0.53 mg/m^3 (ave. 1982-1986), indicating conditions of ultra-oligotrophy according to KEREKES (1983). The lowest values (0.43 and 0.46 mg/m^3) can been found in 1984 and in 1986.

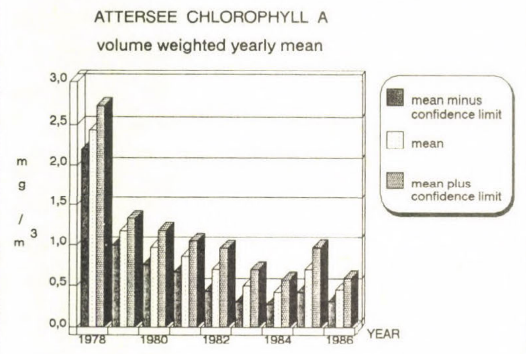


FIG. 6: ATTERSEE - CHLOROPHYLL A - CONCENTRATION

The transparency of the lake water increases with decreasing algal density. Compared to an average transparency of 9.9 m in the period 1978-1981, an average of 11.6 m is measured between 1982 and 1986, whereby the high winter visibilities of 17-18 m are particularly worthy of note (Fig. 7). Maximal Secchi depths occur in spring, minima can be observed during the period of biogenic calcification. Minimal transparency also increases with reduced P-load (from 4.3 m between 1978 and 1981 to 5.7 m between 1982 and 1986). A comparison with early Secchi- depth measurements (DIMITZ 1936) shows that at present Attersee has the same transparency

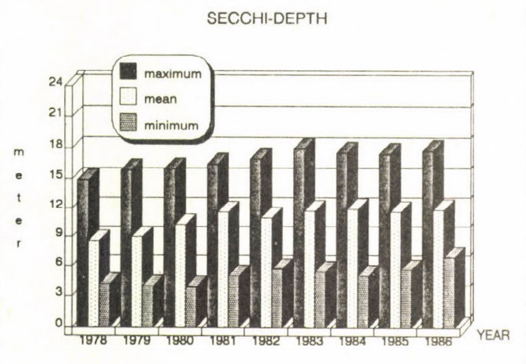


FIG. 7: ATTERSEE - SECCHI - DEPTH

as measured in the year 1935. Although the mean transparency of a hardwater lake is not a suitable trophic indicator because of biogenic calcification, transparency also indicates ultra-oligotrophic conditions according to KEREKES (1983).

The phosphorus concentrations of the lake water, although they decrease slightly, do not significantly reflect the trend of P-loading, chlorophyll, or Secchi-depth measurements (Fig. 8). The volume-weighted yearly averages are between 5-7 mg/m^3. The confidence limits of the mean (due to their similarity all data of a single year have been used) remain within the narrow range of minimal 0.23 and maximal 0.91 mg/m^3 total-P. Considering these low values, the homogeneous P-distribution throughout the investigation period might be due to the fact that the P-concentration values of the lake water also range within the same order of magnitude as the analytical biases. According to GOLTERMAN (pers. comm.) the Ca-concentration can also control the P-concentrations.

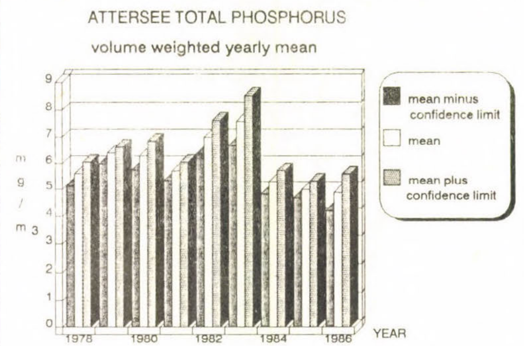


FIG. 8: ATTERSEE - TOTAL PHOSPHORUS - CONCENTRATION

In 1986 only 4.9 mg/m^3 total-P could be measured, indicating conditions of ultra-oligotrophy (VOLLENWEIDER 1971).

This provides evidence that after the reduction of sewage input and the decrease of the nutrient load to about 1/3 of VOLLENWEIDER'S critical load, the water quality of oligotrophic Attersee changes to an ultra-oligotrophic status.

The change from eutrophic to mesotrophic and from mesotrophic to oligotrophic conditions is each known to involve a change in the trophic estimation parameters by a factor of three. The phenomena observed at Attersee allow this scheme to be extended by one level. The change from an oligotrophic to an ultra-oligotrophic state here corresponds to a reduction of the nutrient input to about one third of the critical load: this difference is in the same order of magnitude as that dinstinguishing the other trophic levels.

Acknowledgements - The study is part of the Austrian Eutrophication Programme of the Austrian Academy of Sciences (Coordination: Univ.Prof. Dr. H. Löffler, Vienna).

References:

DANECKER, E. (1969): Bedenklicher Zustand des Mondsees im Herbst 1968.- Österr. Fischerei 22: 25-31.

DIMITZ, L. (1936): Biologische Untersuchungen am Attersee (1934-1936).- Diss. Univ. Wien: 43 pp.

FLÖGL, H. (1976): Die Ringkanalisation am Attersee.- ÖWW 28, 9/10: 186-193.

FLÖGL, H. & A. KOLMHOFER (1976): Die dreistufige Kläranlage Mondsee.- ÖWW 28, 1/2: 17-24.

GÄCHTER, R. & O. FURRER (1972): Der Beitrag der Landwirtschaft zur Eutrophierung der Gewässer in der Schweiz.- Schweiz. Z. Hydrol. 34, 1: 41-70.

HAMM, A. (1976 a): Untersuchungen zur Nährstoffbilanz am Tegernsee und Schliersee nach der Abwasserentfernung - zugleich ein Beitrag über die diffusen Nährstoffquellen im Einzugsgebiet bayerischer Alpen- und Vorlandseen.- Z. Wasser- u. Abwasserforsch. 9: 110-121.

HAMM, A. (1976 b): Zur Nährstoffbelastung von Gewässern aus diffusen Quellen: flächenbezogene P-Abgaben - eine Ergebnis- und Literaturzusammenstellung.- Z. Wasser- u. Abwasserforsch. 9: 135-149.

HASLAUER, J. (1979): Chemische Untersuchungen des Fuschlsees im Jahr 1978 und Nährstoff-Frachtberechnungen.- Arb. Labor Weyregg 3: 53-67.

HASLAUER, J., O. MOOG & M. PUM (1984): The effect of sewage removal on lake restauration (Fuschlsee, Salzburg, Austria).- Arch. Hydrobiol. 101: 113-134.

JAGSCH, A., O. MOOG & M. DOKULIL (in prep): The effect of sewage removal on the limnological development of Mondsee (Austria).-

KEREKES, J. (1983): Predicting trophic response to phosphorus addition in a Cape Breton Island lake.- Proc. N.S. Inst. Sci. 33: 7-18.

MÜLLER, G. (1976): Attersee - Vorläufige Ergebnisse des OECD Seeneutrophierungs- und des MaB-Programms.-Gmunden:179 pp.

MOOG, O. (1982): Selbstreinigende und Phosphorrückhalte-vorgänge in der Seenkette Fuschlsee-Mondsee-Attersee.- Final report of the Austrian Eutrophication Programme, Part I (ÖEP-I), Österr. Akad. Wissenschaften, Vienna: 140 pp.

MOOG, O. (1984): Stream phosphorus exports from prealpine watersheds with respect to geology and landuse.- Verh. Internat. Verein. Limnol. 22: 1071-1076.

MOOG, O. (1987): Gewässerbelastung durch diffusen Nährstoffeintrag unter besonderer Berücksichtigung des atmosphärischen Eintrags und der Niederschlagsversauerung.- Final report of the Austrian Eutrophication Programme, Part II (ÖEP-II), Österr. Akad. Wissenschaften, Vienna: 127 pp.

MOOG, O. (in prep): The diffuse nutrient input by dry and wet deposition within the Salzkammergut lake area (Austria).- Acta hydrochim. et hydrobiol.-

MOOG, O. & G. SCHINDLBAUER (1982): Export coefficients as a tool for nutrient budgeting.- Arb. Labor Weyregg 6: 57-79 (in German).

RECKHOW, K. H., M. N. BEAULAC, & J. T. SIMPSON (1980): Modeling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients.- US-EPA-440/5-80-011.

VOLLENWEIDER, R. A. (1971): Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in Eutrophication. OECD, Paris: 159 pp.

VOLLENWEIDER, R. A. (1975): Input-Output models with special references to the phosphorus loading concept in limnology.- Schweiz. Z. Hydrol. 37,1: 53-84.

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

OFF-FLAVOUR PROBLEM OF LAKE BIWA AND RELATED REMEDIAL STUDIES IN WATER SUPPLY

T. GODA, I. SOHMIYA,* M. TOMITA**

17-8 Ikeda-Nakamachi, Neyagawa, Osaka 572, Japan
*Yoshida-Hommachi, Sakyo-ku, Kyoto 606, Japan
**3 Awataguchi-Kacho-cho, Higashiyama-ku, Kyoto 605, Japan

INTRODUCTION

With the progress of eutrophication of Lake Biwa, which is the supply source of Kyoto Municipal Water Works, an earthy-musty odour has often been experienced by the consumers since 1969. The cumulative research and survey by Kyoto Municipal Water Quality Laboratory has proved the odour-producing micro-organism and the odorous substance to be geosmin by Anabaena macrospora and 2-MIB by Phormidium tenue and Oscillatoria tenuis, respectively. Therefore, the conventional remedy of treating odorous raw water has been the dosing of powdered activated carbon (PAC) at some rate between 5 and 25 mg/l. The PAC dosage rate is determined by indices such as TON values of raw and supplied water, and also the actual concentrations of odorous substances were measured by GC-MS. It is doubtful whether the odorous substances are adsorbed effectively by the PAC particles, and the odour-removal ratio was actually often at an unsatisfactory level. Our effort to solve this kind of off-flavour problem has naturally led us to seek another kind of access, i.e. deodorization using the masking agents.

At first, the quantative relationship between the TON value and the present amount of odorous substance was checked at various temperatures and in the presence of coexisting substances such as chlorine and the trace amount of masking agents. For the above results, secondly, the adaptability of theoretical models - Weber-Fechner and Stevens' Models - was examined and assessed, discussing the changing characteristics of both the degree of sensitivity measured by the organoleptic sensor, S, and the TON values for the different test conditions. Thirdly, the masking effect of food-additive substances, like citral and leaf alcohol, were investigated, and finally, comprehensive mutagenicity tests were carried out inclusively for both the odorous substances (geosmin, 2-MIB) and the masking agents. This mutagenicity test includes the Ames test and the umu-test which has been developed recently.

Other studies, like those on the fate and behaviour of masking agent in the water supply system are still going on in order to grope the way of practical application of such a masking method.

EXPERIMENTAL STUDIES

TON values for various water samples were measured by two panelists, based on the Japanese Standard Methods for Examination of Supplied Water (JWWA 1985, revised). The odorous substances used in this test were: synthesized geosmin (98% pure, Wako Co.) and dimethylisoborneol (99.9% pure, Wako Co.). These substances were, first, diluted by distilled water to obtain a substance concentration of a level between 20 and 800 mg/l. These were used as standard solutions. The standard samples were then examined to define TON values for both warm (40~50^{o}C) and cool (11~15^{o}C) conditions. Here, the two panelists were both experts who were accustomed to performing this kind of test.

On the other hand, accurate concentrations of odorous substances were ascertained by a GC-MS (Shimazu QP-1000) analyzer. Analyses were made for the following three cases:

1. adding 1 mg/l and 2 mg/l chlorine;
2. diluting solution with raw, conducted lake water;
3. adding some aromatic, food-additive substances (citral and cis-3-hexen-1-ol).

Besides, we checked the differences in the measured TON values for winter and summer, using the same standard solutions; the results seemed reasonable. The experimental conditions are summarized in Table 1.

Table 1. Test condition

Odorous substances only (standard sample)

	Geosmin	2-MIB	Geosmin + 2-MIB
Warm	RUN-A	RUN-B	RUN-C
No. of data	15	15	15
Cool	RUN-A′	RUN-B′	RUN-C′
No. of data	10	10	10

Presence of coexisting materials

	Geosmin				2-MIB			
	Chlorinated water		Raw water (dil.)	Citral or leaf al.	Chlorinated water		Raw water (dil.)	Citral or leaf al.
	1 mg/l	2 mg/l			1 mg/l	2 mg/l		
Warm	RUN-D_1	DUN-D_2	RUN-E	RUN-F	RUN-G_1	RUN-G_2	RUN-H	RUN-I
No. of data	15	15	15	6	15	15	15	6
Cool	RUN-D_1′	RUN-D_2′	RUN-E′	RUN-F′	RUN-G_1′	RUN-G_2′	RUN-H′	RUN-I′
No. of data	12	15	14	6	15	15	14	6

The reasons for selecting the above three cases for test are as follows:

– Chlorine is in world-wide use as the disinfectant of waterworks.

– Various organic and inorganic substances are found in both raw and supplied water.

– Citral and cis-3-hexen-1-ol (leaf alcohol) are both considered as representative masking substances for such a case.

The citral used is of 95% purity (made by Wako Co.) and the leaf alcohol is 98% pure (made by Aldrich Chemical Co).

RESULTS AND CONSIDERATIONS

1. Adaptability assessment of theoretical models. Fifteen data were obtained for each experimental condition. These are plotted in several figures (and can be found in "Water Science

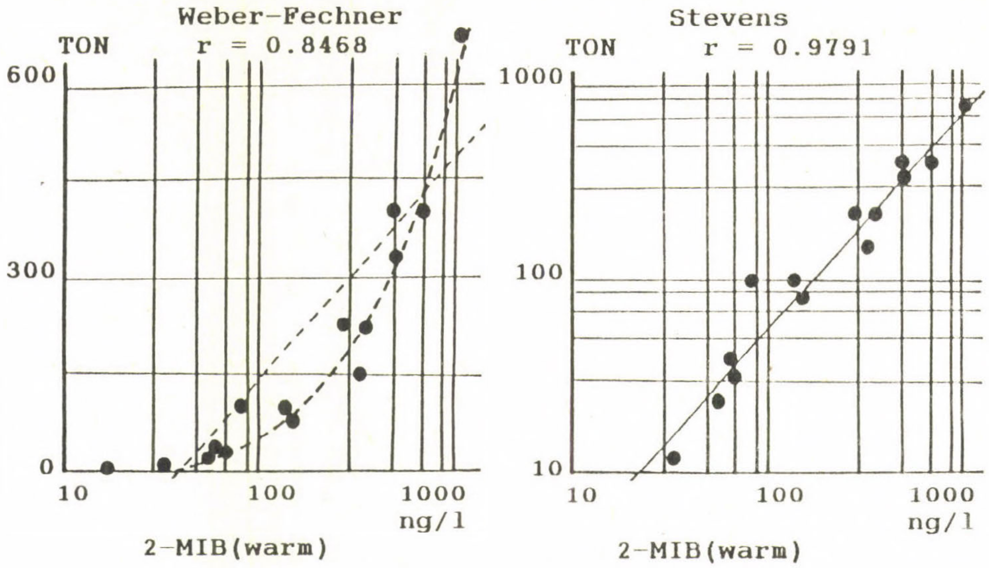


Fig. 1. Comparison of the Weber-Fechner and Stevens models

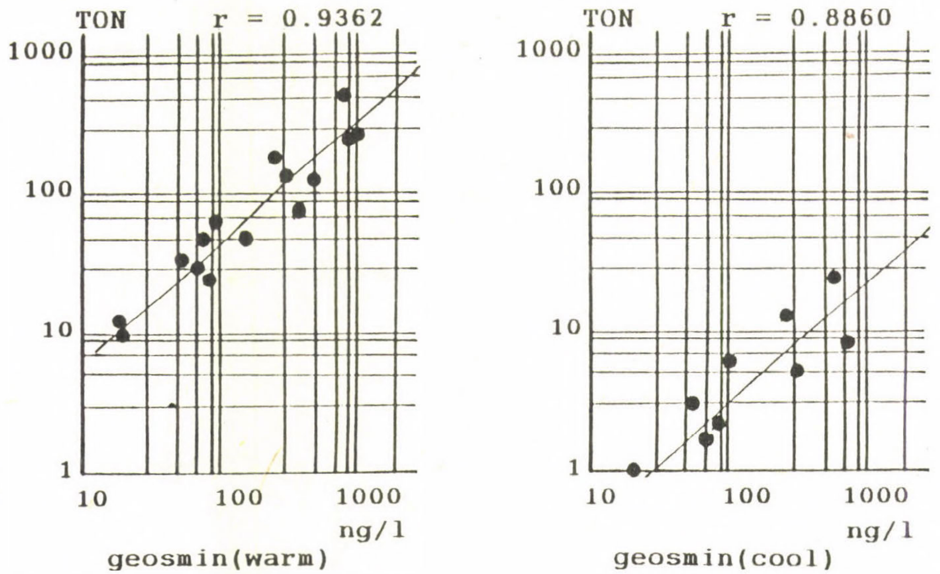


Fig. 2. Effect of temperature

and Technology") (1), proving that they fit rather to the Stevens Model than to the Weber-Fechner Law; i.e. the relation of TON versus concentration on the log-log paper is scattered along the straight line (Fig. 1).

2. Effect of temperature. As for the data of standard samples, fitting to the Stevens Model seems to be better for the case of warm conditions than for the cool ones, yet, the degree of sensitivity, in other words, the slope of the straight line on the log-log paper is much steeper for the data of the warm group. In case there are some coexisting substances, the slope becomes generally steeper, as compared to the case of the standard solution independent of the temperature. Furthermore, TON values were about one order of magnitude lower for cool than warm conditions (Fig. 2).

3. Effect of chlorine. By adding 2 mg/l of chlorine, TON decreases to about half of the initial value (standard solution) (Fig. 3).

4. Coexisting effect of masking agents. By adding ca. 0.01 mg/l of citral or leaf alcohol, TON values had decreased by about one order of magnitude from the standard conditions (Fig. 4). By this, we can expect these substances to actually mask the earthy-musty odour originating from geosmin and dimethylisoborneol under purer condition.

5. Toxicity and mutagenicity of odorous substances and masking agents. When we consider this kind of masking method for practical use, both of the original odorous substances and masking agents should be assessed for their risk on the consumer's health. Hence, at a first step we checked either the toxic effect or the mutagenicity. The toxicological data were collected from the relevant references. It should be noted that citral and leaf alcohol are both appointed food additives authorized by the Japanese Ministry of Health and Welfare, so they are widely used for cooking and other purposes.

The results of the referential survey are summarized in Table 2. Considering the anticipated dose of these masking agents, it seems that there may be no problem in view of the acute and chronic toxicity.

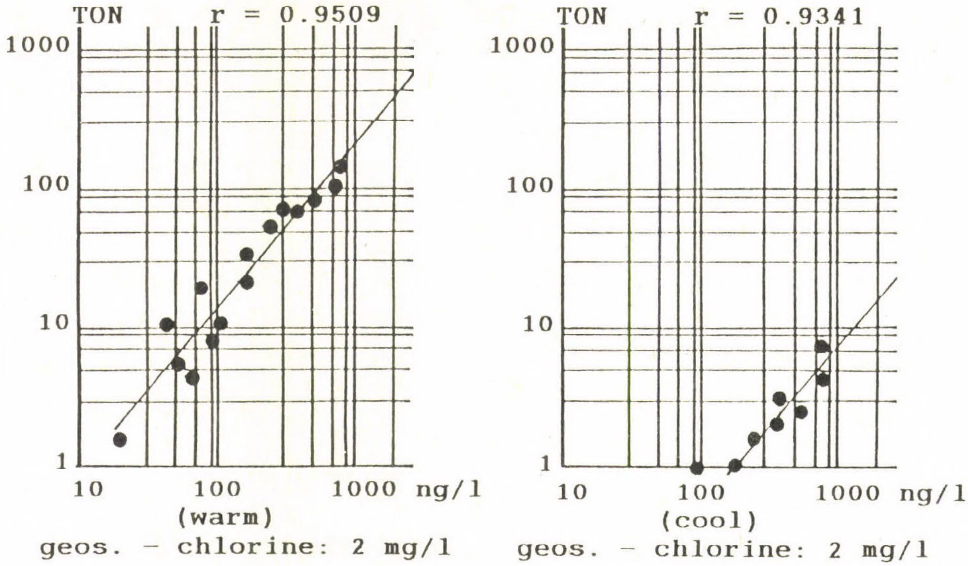


Fig. 3. Effect of chlorine

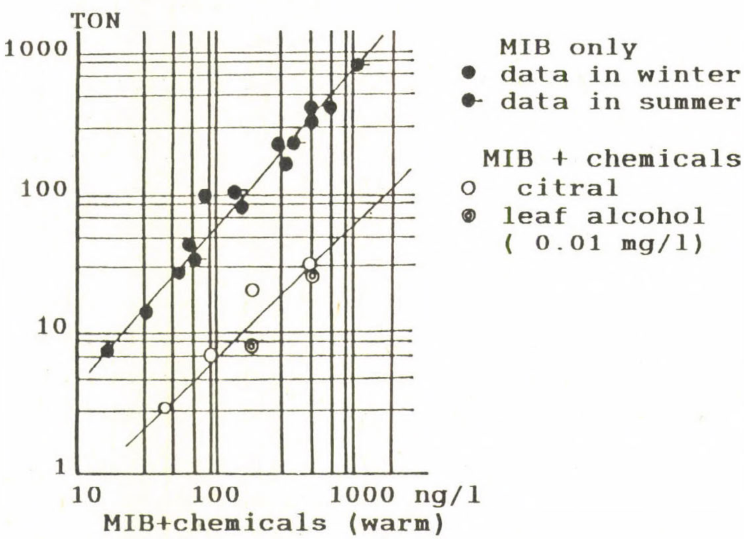


Fig. 4. Effect of citral and/or leaf alcohol

Table 2. Toxicity data of masking agents (citral : leaf alcohol)

	Rat		Mouse	
	orl	ipr	orl	ipr
Citral	LD50 4960 (mg/kg)	*	LD50 6000 (mg/kg)	*
Leaf alcohol	LD50 4700 (mg/kg)	LD50 600 (mg/kg)	LD50 7000 (mg/kg)	LD50 400 (mg/kg)

Table 3. Example of comparing sensitivity, specificity and accuracy in both Ames and umu-test for various carcinogens (from Nakamura, S. et al., Mutation Res. 192, 239, 1987)

	umu-test	Ames test
Sensitivity	0.69 (54/78)	0.74 (58/78)
Specificity	0.87 (20/23)	0.86 (19/22)
Accuracy	0.73 (74/101)	0.77 (77/100)

umu-test is rather simple and reliable.

The mutagenicity of odorous substances is mainly confirmed by the umu-test, which is a rather simple method developed recently by Shinagawa et al. (2). Reliability of the umu-test is seen in Table 3, showing that the major characteristics of the umu-test are quite similar to those of the Ames test. Table 4 demonstrates that four substances (geosmin, 2-MIB, citral and leaf alcohol) are negative in mutagenicity for the practical range of concentration, 1 ppm ~ ppt. These results have been reconfirmed by the Ames test, too.

FUTURE PROBLEMS AND CONCLUSION

We have introduced a new approach of controlling off-flavour problems in water supply, using some masking agents which are not synthesized but of natural origin. Prior to de-

Table 4. Mutagenicity estimation
(by umu-test on odorous substances and masking agents)

		Concentration of substances (mg/l)						
Subs.	blan.	10^{-6}	10^{-5}	10^{-4}	10^{-3}	10^{-2}	10^{-1}	10^{0}
2-MIB	a_1	(-)	(-)	(-)	(-)	(-)	(-)	(-)
Geosmin	a_2	(-)	(-)	(-)	(-)	(-)	(-)	(-)
Citral	a_3	*	*	*	(-)	(-)	(-)	(-)
Leaf al.	a_4	*	*	*	(-)	(-)	(-)	(-)
AF-2	a_5	*	*	*	*	(+)	(+)	*

Notes:

1. Original concentration of substances:
   2-MIB 200 mg/l
   geosmin 80 mg/l
   citral 3000 mg/l
   leaf alcohol 3000 mg/l

2. AF-2: Furylfuramide

3. Method of estimation:
   $E = (x_i - a_i)/ai$
   x_i = β-galactosidase activation value under each substance concentration
   a_i = β-galactosidase activation value under the blank of each substance
   (-): $<E$ 0.5
   (+): $>E$ 1.0

veloping such a method in practice, many things have to be clarified, e.g. the identification of risk levels of odorous substances, physical and chemical properties of masking agents, biodegradability and resisting characteristics of masking agents against chlorine and biofilms.

On the other hand, the response of the consumer and the acceptability of the related authorities should, of course, be investigated in detail. Before proceeding to the practical application of the masking method in waterworks, the following items have remained for future study:

1. Trend of TON values according to the dosage of masking agents under conditions of the coexistence of various substances and microorganisms.

2. Accumulation of experimental data either for the bench-scale system or for the actual, model-water distribution system.

3. Survey of the responses of the relevant agencies, citizens and water consumers in applying the masking method for the control of the off-flavour problem.

4. Finding of a more favourable masking agent concerning performance and durability in the water system and economy.

REFERENCES

1. M. Tomita, N. Ichikawa, T. Goda: Correlation Analysis of TON and Actual Presence of Odorous Substances in Water Supply. Preprint of the Second International Symposium on Off-Flavours in the Aquatic Environment, 20, Kagoshima, Japan (Oct. 1987).
2. H. Shinagawa et al.: Preprint of the 11th Annual Meeting of the Japanese Society on Environmental Mutagen, 38 (1982).

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

UTILIZATION OF LAGOON ECOSYSTEM IN NUTRIENT REMOVAL FROM THE INFLOWING WATERS TO THE LAKE

A. KURATA

Lake Biwa Research Institute
1-10, Uchide-hama, Otsu, Shiga Prefecture, 520, Japan

INTRODUCTION

Lake Biwa is the largest freshwater lake in Japan and is located in the center of the main island. Recently eutrophication has proceeded extensively in the lake and has caused serious trouble for people receiving their water supply from this lake. In order to reduce problems caused by eutrophication, various ordinances have been enacted and many improvement works have been made by responsible authorities. However, eutrophication still progresses slowly but steadily by the increase of nutrient loading from a number of sources, especially non-point sources such as domestic wastewater and cultivated field runoff. Nowadays, therefore, removal of nutrients from the water inflowing to the lake is one of the most important approaches to solving the environmental problems of the lake.

DESCRIPTION OF STUDY AREA

There are many large and small natural lagoons around the lake. These lagoons are very shallow, approximately 1 or 2 m in depth, connecting with the lake through narrow channel with the same water level. The lagoons are classified into five types as shown in Table 1 and play a significant role in removal of nutrients loading to the lake as reported in an earlier paper by Kurata (1983).

Phragmites communis grows well in Nishinoko Lagoon which is the largest lagoon around the lake. The grown area of P. com-

Table 1. Category of five types of lagoon around Lake Biwa

	Lagoon				
	I	II	III	IV	V
Sport-fishing	x	x	x	x	
Reed zone	x	x		x	
Freshwater pearl cultivation farm	x		x		x
Freshwater fish cultivation farm		x			

munis covering about 167 ha is shown in Fig. 1. Sampling stations were set up linearly from the main inlet to the outlet as shown in Fig. 2.

RESULTS AND DISCUSSION

As shown in Fig. 3, a great amount of inorganic and organic particulate matter contained in the inflowing water was sedimented in the lagoon, within several hundred meters from the inlet. After flow into the lagoon, the temperature of influent was raised about several centigrades in the lagoon and it must accelerate the activity of mineralization by very high population density of heterotrophic microorganisms compared with that in the lake water. It is supposed that the dissolved-form nitrogen compounds must be reduced through denitrification in the reed zone by nitrate and nitrite reducers as shown in Fig. 4.

During the winter, P. communis is harvested annually and it is used for different kinds of daily life products, for example, sunshade screen, domestic screen, fancy goods, etc. Approximately 910 t (dry wt.) of P. communis is harvested annually from this lagoon. Accordingly, the same amount of organic matter is removed annually from this ecosystem. Nitrogen and phosporus contents contained in P. communis were assayed and the total amount of nitrogen and phosphorus removed from the lagoon was calculated. The results are shown in Table 2. Past

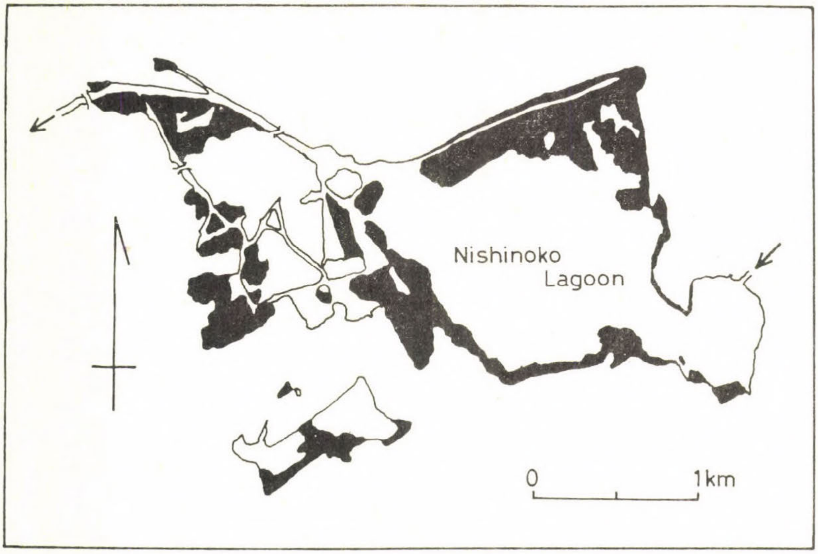


Fig. 1. Profile of Nishinoko Lagoon which is the largest lagoon around Lake Biwa. Arrows indicate the main inlet and the outlet. Solid parts represent reed zone.

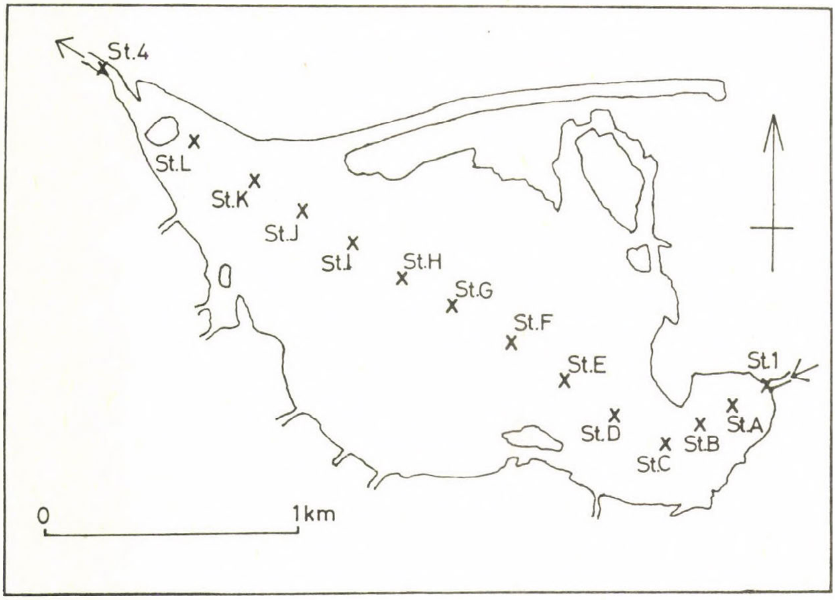


Fig. 2. Location of sampling stations set up linearly in the lagoon.

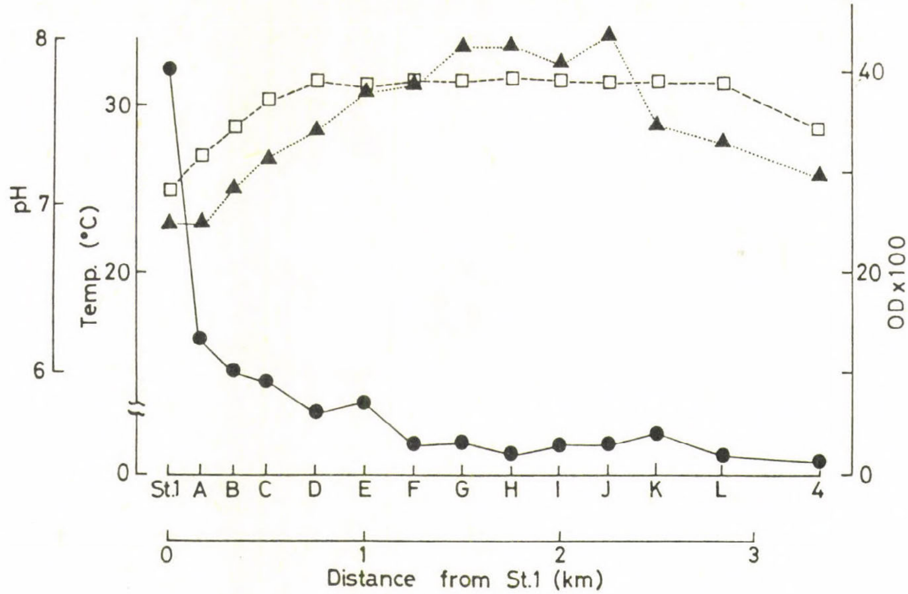


Fig. 3. Successive variations of water temperature, pH value and turbidity from St. 1 (inlet) to St. 4 (outlet). □---□ water temperature; ▲···▲ pH; ●——● optical density (640 mμ).

trend of forwarding of P. communis products is shown in Table 3.

The shellfish Hyriopsis schlegeli is cultivated to produce freshwater pearls in the lagoon. Distribution of the cultivation farms for pearl production is shown in Fig. 5. The larvae of the shellfish are also collected in the central part of this lagoon.

Surrounded by paddy fields, the lagoon receives very high nutrient load. However, it is thought that the activity of mineralization by heterotrophic microorganisms is very high both in water and in the surface of bottom sediments in the lagoon. It enables to sustain the very high population density of phytoplankton, 100 times or more, compared with that in the lake. As a token of its phenomenon, a very close relationship is found between the population density of phytoplankton and the conversion ratio from the dissolved-form inorganic phosphate to the particulate-form organic phosphate.

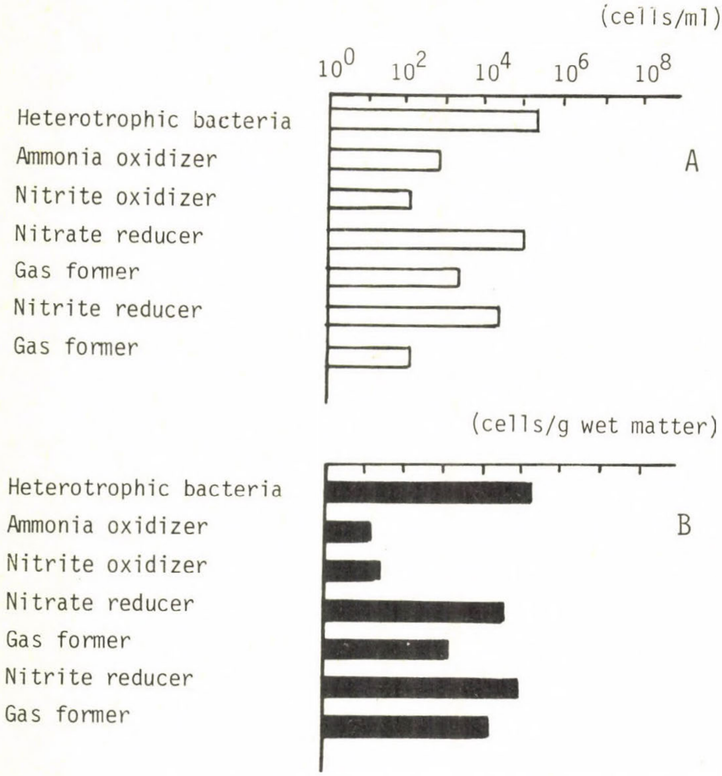


Fig. 4. Bacterial population density having various functions in water (A) and in the bottom sediments (B) in the lagoon.

A high biomass of phytoplankton is consumed for the growth of H. schlegeli, namely, for the production of freshwater pearls. In this lagoon, any fertilizer is not used for acceleration of the growth of phytoplankton and also any bait is not given to the shellfish.

Accordingly, it is concluded that loaded nutrients from the surrounding cultivated fields and from upstream villages and towns are converted to freshwater pearls by a number of H. schlegeli, although requiring skilful technique, troublesome work and delicate management of environmental conditions of cultivation farms.

It is estimated that approximately 700,000 H. schlegeli are harvested annually from the lagoon. After picking up pearls

Table 2. Annual removal of nitrogen and phosphorus from the lagoon by the harvest of *Phragmites communis*

Total area of *P. communis*	60 ha
Average number of *P. communis*/m^2	52
Nitrogen content in *P. communis*/m^2	36.6 g
Phosphorus content in *P. communis*/m^2	4.3 g
Total harvest of *P. communis* (dry wt.)	910 t
Total nitrogen removal	16.4 t
Total phosphorus removal	2.3 t

Table 3. Past trend of forwarding of *Phragmites communis* products

Year	No. of factory	Employee	Sum of forwarding (10^3 yen)
1974	17	64	178,600
1975	19	73	197,380
1976	20	103	179,710
1977	16	71	201,850
1978	16	100	226,820
1979	15	77	202,140

from the shellfishes for about five years' cultivation, they are usually fit for consumption.

These processes are effective in removal of nutrients from the lagoon. Past trend of freshwater pearl production in the lagoons around Lake Biwa is shown in Fig. 6. In 1970, approximately 7 t of the total amount of freshwater pearl was produced in Shiga Prefecture. However, its production is decreasing year by year because of the deterioration of the water quality of cultivation farms. Approximately 40% of freshwater pearl produced in the prefecture is produced in Nishinoko Lagoon. Most of it is being exported.

The submerged macrophytes grow well in the open area of this shallow, and nutrient-rich lagoon. During the summer, therefore, macrophytes had been harvested by many fishermen to

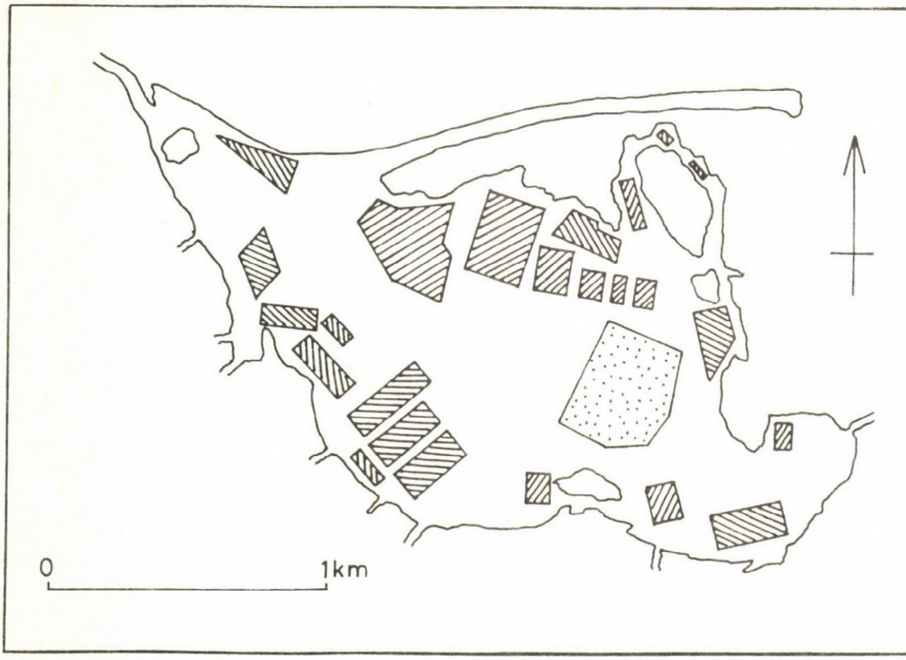


Fig. 5. Freshwater pearl cultivation farms (parts of oblique lines) in the lagoon. The dotted part represents the area from which the larvae of Hyriopsis schlegeli are collected.

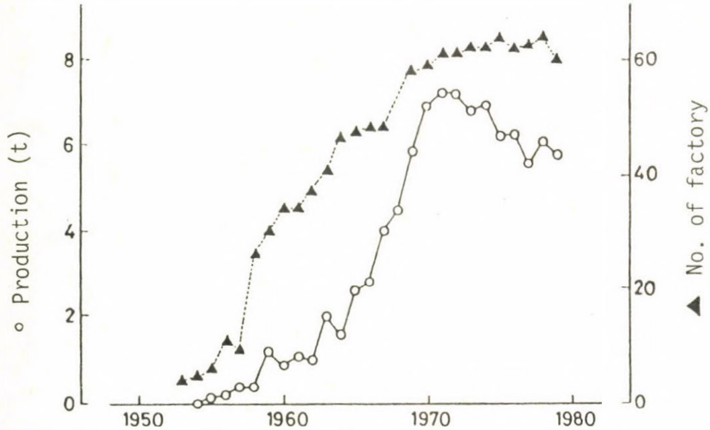


Fig. 6. Past trend of freshwater pearl production in the lagoons around Lake Biwa.

Table 4. Removal of nitrogen and phosphorus from the lagoon by various human activities

	Total nitrogen (t)	Total phosphorus (t)	Value of annual harvest (10^6 yen)
Harvest of *Phragmites communis*	16.4	2.3	180
Harvest of *Hyriopsis schlegeli*	4.0	0.27	1,500
Harvest of *Elodea canadensis*	10.5	1.5	-
Fish catch	0.3	0.02	8
Total	31.2	4.09	1,688

improve environmental conditions for navigation and cultivation of freshwater pearls in the lagoon. In 1984, approximately 3,000 t (wet wt.) of *Elodea canadensis* was harvested and used as a manure to the orchard. Since the late summer of 1984, an automatical macrophyte harvesting vessel was introduced into the lagoon.

In addition to this process, sport-fishing equivalent to 8 million yen is carried out throughout the year.

Finally, some kinds of effective processes for the removal of nutrients from this ecosystem are summarized in Table 4. Thus harvesting of nitrogen and phosphorus contributed 38.5 and 93.0%, respectively, to the entire nutrient retention of the lagoon.

Nishinoko Lagoon plays an important role as a whole ecosystem in removing nutrients from the inflowing wastewater into Lake Biwa, as if it were a natural ecological wastewater treatment system.

REFERENCE

Kurata, A. 1983. Nutrient removal by epiphytic microorganisms of *Phragmites communis*. In. R.G. Wetzel (ed.), Periphyton of Freshwater Ecosystems. Dr. W. Junk Publishers, The Hague, pp. 305-310.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest
Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

PHYTOPLANKTON SUCCESSION IN A LARGE WATERBODY

L.A. PAUTOVA, V.N. PAUTOVA, V.A. SILKIN

Institute of Ecology of the Volga River Basin,
Academy of Sciences of the USSR
445003, Togliatti, USSR

When studying the human-induced eutrophication of waterbodies it is important to reveal the reasons causing alterations in a seasonal succession of planktonic phytocenoses under the impact of an increasing nutrient load.

A certain scheme of seasonal successions caused by varying nutrient concentrations has been recently elaborated for marine ecosystems (Margalef, 1984). Margalef distinguishes three phases in a seasonal succession. The first phase is concerned with the growth of small diatoms at a relatively high nutrient concentration under homothermic conditions. The second phase is characterized by the increase in density of large diatoms and decrease in density of small ones with the start of stratification. In an undisturbed ecosystem of a marine offshore zone as well as in pollution-free freshwater reservoirs and lakes (Reynolds, 1984) in the presence of stratification excluding any nutrient release from bottom sediments, common for the third phase is the development of large dinoflagellates at the background of their high species diversity and low total phytoplankton density. The length of each phase varies under the impact of a steady inflow of nutrients. In the presence of a continuous mixing, for example, the length of the first phase tends to increase, and that of the second and third ones to decrease. In eutrophication-impacted marine off-shore zones small diatoms or Pyrrophyta are dominant during the whole vegetation period, i.e. the first succession phase can be permanently observed.

In freshwater eutrophic waterbodies planktonic succession proceeds as follows: small diatoms --- large diatoms --- green and bluegreen algae (Reynolds, 1984; Triphonova, 1986). Results of monthly field observations on the Kuibyshev Reservoir in 1975-1984 show that in spring there occurs the development of diatoms from genera Stephanodiscus, Melosira, Cyclotella. A lot of small non-identified diatoms (plausibly from genera Stephanodiscus and Cyclotella) can also be found. In late spring and early summer, in the presence of an increasing growth of diatoms, abundant are green protococcal (and in recent years pyrrophitic) algae - cryptomonades Chroomonas acuta Üterm., Cryptomonas caudata Schiller, Cr. rostrata Troitzkaja emend I. Kissel). In summer they are succeeded by bluegreen algae in the presence of remaining high density of diatoms and green algae. The dominance of bluegreen algae in lake-shaped enlargements during the autumn period still remains, and sometimes there are instances of high densities of Melosira and Stephanodiscus. Complexes of spring and early-summer phytoplankton in different years vary in species composition.

Experiments with algal cultures enable to explain the algal succession in a given large waterbody, say, the Kuibyshev Reservoir. In eutrophic waterbodies where there are practically no shortages in nutrient supplies, particularly in N and P, their ratios are of paramount significance for phytoplanktonic succession. In mixed cultures consisting of diatoms, green and bluegreen algae the former two are known to demonstrate a faster growth at high Si:P ratios (100:1 and more). Chlorophyta are dominant at high N:P ratios and Cyanophyta at low (Tilman, Kiesling, 1984).

Laboratory experiments based on a "group culture" method (Løvstad, 1984) were carried out using phytoplanktonic samples taken from the Kuibyshev Reservoir. A natural phytoplanktonic population isolated from the reservoir waters in November 1987, was adapted to an artificial habitat Chu-10. The adaptation process lasted for a month and a half at 8-11°C and illumination of about 9 W/m^3 provided by a LB-30 bulb. The resulting mixed culture consisted of Cyclotella glomerata, Diatoma elongatum, Ankistrodesmus pseudomirabilis, Chlorella sp., Navicula

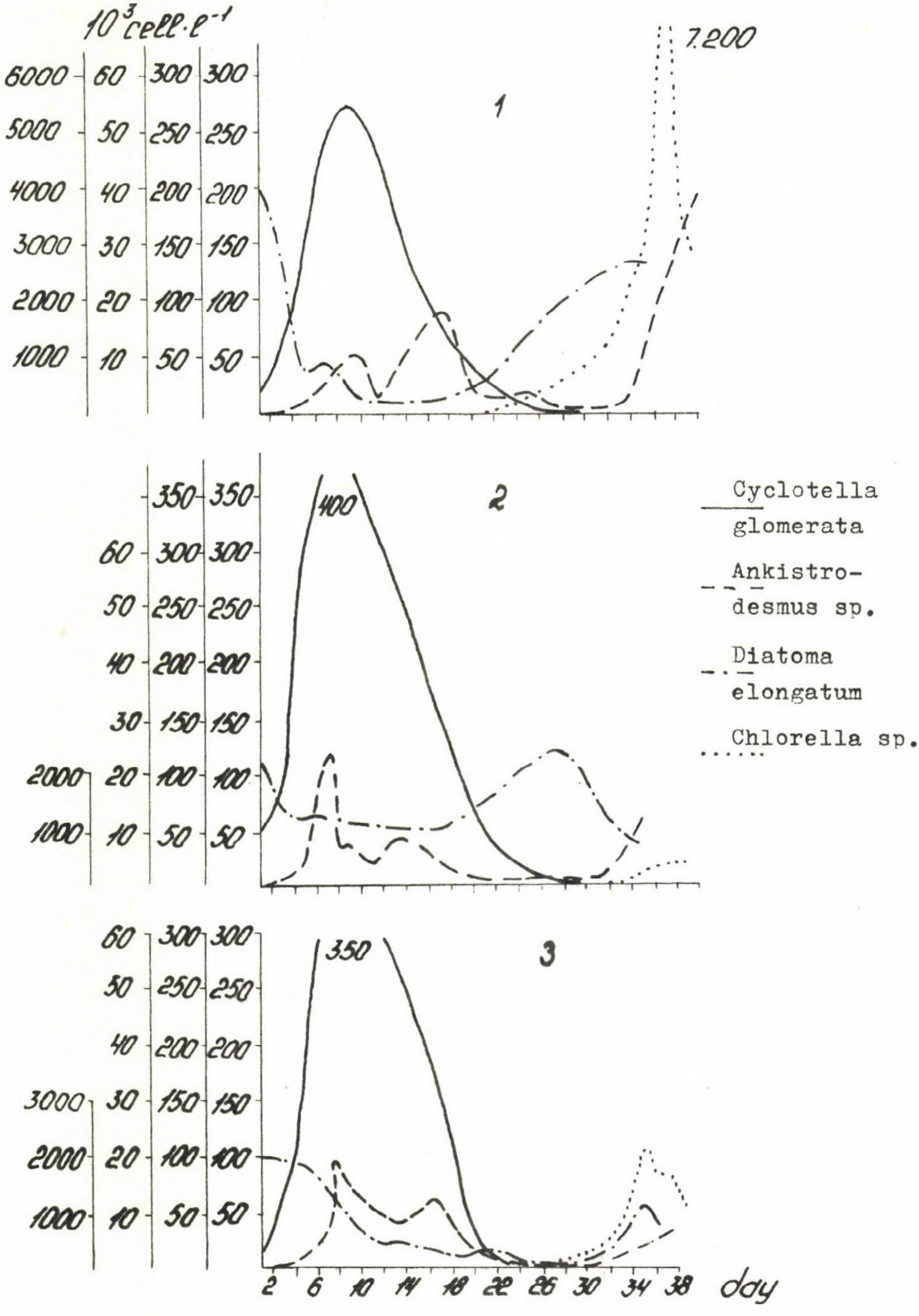


Fig. 1. Dynamics of basic species of spring phytoplankton in the Kuibyshev Reservoir, as experimented with Chu-10 habitat: 1 - control; 2 - 0.5 N; 3 - 0.5 P

sp. The former two are typical to the dominant complex of spring-summer phytocenoses both in the Kuibyshev Reservoir and other impoundments with a moderately eutrophic state. The subsequent 38-day experiment allowed a simulation to be made of a succession in a spring-and-early summer phytocenosis from small Cyclotella glomerata to large Diatoma elongatum and further on to green algae. Available was a chance to observe a classical succession of species (Margalef, 1984; Reynolds, 1984) presented in Fig. 1.

Our objective was to reveal the impact of inorganic N:P ratio on phytoplanktonic seasonal successions. Therefore, apart from a complete growth medium Chu-10 with a double concentration of silicon, two more modified versions were also used throughout the experiment, the first one containing a halved nitrogen and a full phosphorus concentration and the second a halved concentration of phosphorus and a full concentration of nitrogen. Each medium was run three times. It was established that reduction in N and P concentrations does not significantly impact the succession of species. Deviations revealed were mainly of a quantitative character. Large Diatoma elongatum under the conditions of adequate Si supply was found to grow faster at a lower N:P ratio, while small diatom C. glomerata and green Scenedesmus are known to be insensitive to N:P changes (either halved or doubled). Small green Chlorella was found to be extremely sensitive to N concentration changes and practically ceases its growth at its halved value.

As the length of vegetation period and population density levels are governed by the nutrient concentrations available (Lund, 1965; Rodhe, 1948; Triphonova, 1986), experimental studies of the impacts of given concentrations in addition to permanent and frequent (at least once a week) field observations on the process of natural successions are supposed to be the most perspective way of comprehending the mechanisms of phytoplankton succession.

References

Løvstad Ø., 1984. Competitive ability of laboratory batch phytoplankton population at limiting nutrient levels. Oikos, V.42, N2, p. 176-184

Margalef R., 1984. Le plankton de la mediterranée. La Recherche, V. 15, N58, p. 1082-1094

Reynolds C.S., 1984. The ecology of freshwater phytoplankton. London: New York: New Rochelle: Cambridge University Press, (Ed. E.Beck, H.I.B. Birks, E.F. Connor)., p. 364

Sommer U., 1985. Seasonal Succession of Phytoplankton in Lake Constance. Bioscience, V. 35, N6, p. 351-357

Tilman, D., and R.L.Kiesling. 1984. Freshwater algal ecology: taxonomic trade-offs in the temperature dependence of nutrient competitive abilities. p.314-319 in M.J. Klug and C.A. Reddy, eds. Current Perspectives in Microbial Ecology. American Society for Microbiology, Washington, DC.

Triphonova I.S. 1986. Seasonal and general successions in lake phytoplankton. Gidrobiol. J., V.22, N3, p.21-28

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

STUDIES ON HYDROBIOLOGICAL PROCESSES IN ENRICHED LAKE ENCLOSURES

J. NÉMETH, P. GULYÁS, L. KONCSOS, L. VÖRÖS*

Research Centre for Water Resources Development
VITUKI, H-1453 Budapest, Pf. 27, Hungary
*Balaton Limnological Research Institute,
Hungarian Academy of Sciences, H-8237, Tihany, Hungary

ABSTRACT

The effects of plant nutrients (PO_4-P, NO_3-N) and environmental factors (irradiation, temperature,wind) on the water quality in Lake Balaton, with special emphasis on eutrophication processes were studied in Keszthely bay during the summer period of 1986 and 1987 in four series of enclosure experiments.

The temporal changes of phytoplankton biomass was described by a modified BEM model (Balaton Eutrophication Model) in which the predator-prey relationship of zooplankton and phytoplankton was taken into account.

The experimental results revealed that in the most eutrophicated region of Lake Balaton the effect of enrichment was less expressed due to the autochtonous nutrient loading from the sediment, while in the mesotrophic region (Tihany) the increase of phytoplankton biomass was considerably higher at similar loading rates.

In Keszthely bay algal growth is limited by self-shading and by the grazing effects of zooplankton.These limitations should be considered in modelling eutrophication processes.

The coexistential pattern dynamics of planktonic communities were taken also into consideration.

INTRODUCTION

In the summer periods of 1986 and 1987 enclosure experiments were performed in the Keszthely Basin of Lake Balaton to study the effect of plant nutrients and sediment exchange on the metabolic and structural processes in the aquatic ecosystem. Similar experiment has been completed at Tihany during 1984 (Istvánovics et al. 1986).

The limnocorral equipment (enclosure) developed at VITUKI was used in these experiments. The volume of the water enclosed and the depth ranged from 12 to 17 m^3 and 170 to 200 cm, respectively.

The necessary amounts of the KNO_3 and KH_2PO_4 standard solutions were added and mixed with the water before sampling and in situ measurements, normally between 10 and 12 a.m.

In two periods of the 1986 experiment different nutrient loading rates were used. Two experiments were run in 1987. The experimental conditions are summarized in Table 1.

The state variables studied in situ and in the laboratory included the light conditions (instrumental light intensity and Secchi transparency measurements), dissolved oxygen, water temperature, suspended solids, chlorophyll-a, primary production, nitrogen forms (NH_4^+, NO_2^-, NO_3^-, Kjeldahl-N), phosphorus forms (dissolved reactive PO_4-P, total P), pH, specific electrical conductivity, pupulation density and biomass of phytoplankton and zooplankton.

Global irradiation and wind speed data were obtained from the National Meteorological Service.

The enclosure experiment in the year of 1986 has been made in collaboration with the Balaton Limnological Research Institute, Hungarian Academy of Sciences.

The data involved in the metabolic model (light conditions, suspended solids, water temperature, global irradiation, wind speed, plant nutrients, chlorophyll-a) and the results of the studies on the structure of phyto- and zooplankton are considered more in detail here.

METHODS

Of the state variables considered in the present paper the following were measured in situ:

- Secchi transparency,

Table 1. Applied treatments during 1986, 1987 in the enclosures

Duration	No.of the enclosure	Nutrient loading ($mg\ m^{-3}\ d^{-1}$) NO_3-N	PO_4-P	
July 3-24, 1986 (first period of experiment)	1	0	0	(control)
	2	50	0	
	3	20	5	
	4	100	5	
July 25-August 8,1986 (second period of experiment)	1	0	0	(control)
	2	50	0	
	3	30	15	
	4	250	30	
July 2-20, 1987 (first experiment)	1	0	0	sediment replacement*
	2	0	0	(control)
	3	20	5	
	4	100	5	
July 22-August 10,1987 (second experiment)	1	0	0	water replacement
	2	0	0	(control)
	3	20	5	
	4	100	5	

Notes: * Sediment dredged from 4-5 m depth in the Keszthely basin spread on the bottom

- light intensity (at the surface and at points spaced 25 cm below the surface using the Kahlsico field instrument).

The chlorophyll-a was determined photometrically following methanol extraction (Felföldy, 1987) from the sample representing the average of the water column. The concentrations of the nitrogen and phosphorus forms were determined according to Felföldy (1987).

The population density and biomass of the phytoplankton were determined using an Utermöhl inverted microscope on samples fixed in situ with Lugol solution and conserved with formol.

For quantitative analyses on the zooplankton 5 dm^3 water samples were passed through a No.25 plankton net.The samples were conserved at the site with formol. The Utermöhl inverted microscope was used to determine the population density. The dry weight per unit water volume of the various populations was calculated using the specific values published in the literature (Botrell et al. 1976).

RESULTS AND DISCUSSION

A detailed report on the experimental results will be omitted because of space limitations. The data on global irradiation, wind speed and variations in the phyto- and zooplankton biomass are presented in Figs1,2.

The model taking into account phyto- and zooplankton interactions will be used subsequently to describe the short-term variations in the amount of chlorophyll-a (phytoplankton biomass), whereafter the variations in the coexistential pattern of plankton communities will be analysed by a multivariate method.

Short-term variations in the phytoplankton biomass

The factors and processes involved in the model describing the short-term variations in the phytoplankton biomass are reviewed in Fig. 3.

The equations describing the time variations in phytoplankton biomass

$$\frac{dA}{dt} = \sum_i (G_i - D_i)$$: the basic equation of time variations in the phytoplankton biomass (1)

$$G_i = P_i^* \, FP_i FN_i FL_i FT_i \, A_i$$: the growth rate (2)

$$D_i = D_i^* \exp(\beta_i (T - T_{ci})) A_i$$: the mortality rate (3)

$$FP_i = P/(P_{Oi} + P)$$: factor of phosphorus limitation (4)

$$FN_i = N/(N_{Oi} + N)$$: factor of nitrogen limitation (5)

factor of temperature limitation:

$$F\,T_i = \frac{T_{ci} - T}{T_{ci} - T_{Oi}} \exp\left(1 - \frac{T_{ci} - T}{T_{ci} - T_{Oi}}\right), \quad \text{if} \quad T > T_{ci} \qquad (6)$$

$$F\,T_i = 0, \quad \text{if} \quad T < T_{ci}$$

factor of light limitation:

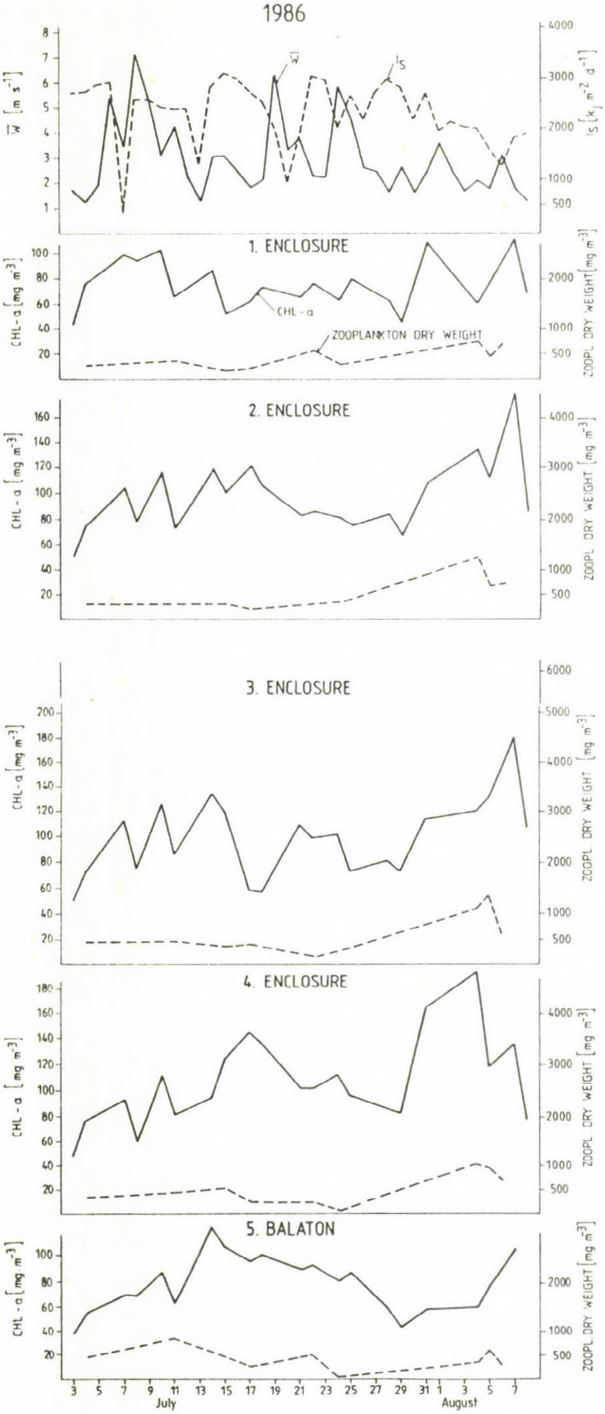


Fig. 1.
Temporal variations of global radiation (I_S), daily mean wind speed ($\bar{W}$), chlorophyll-a concentration (CHL-a), zooplankton dry weight (Z) in 1986

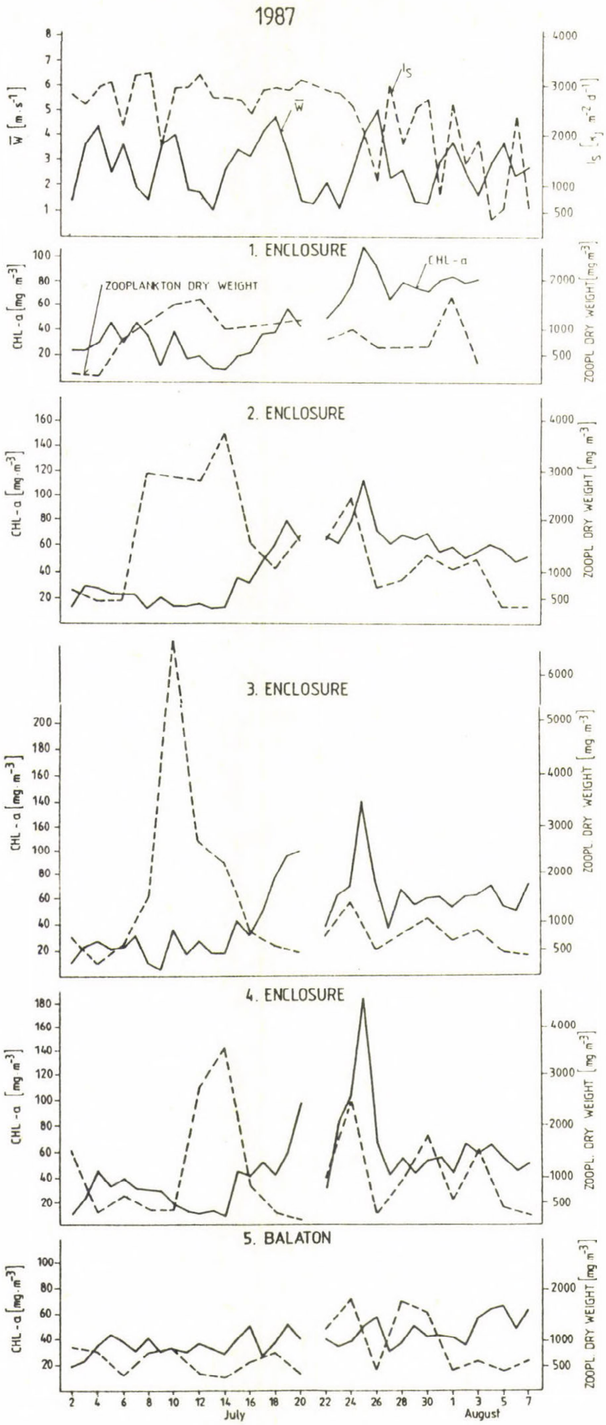


Fig. 2.

Temporal variations of global radiation (I_S), daily mean wind speed ($\bar{W}$), chlorophyll-a concentration (CHL-a), zooplankton dry weight (Z) in 1987

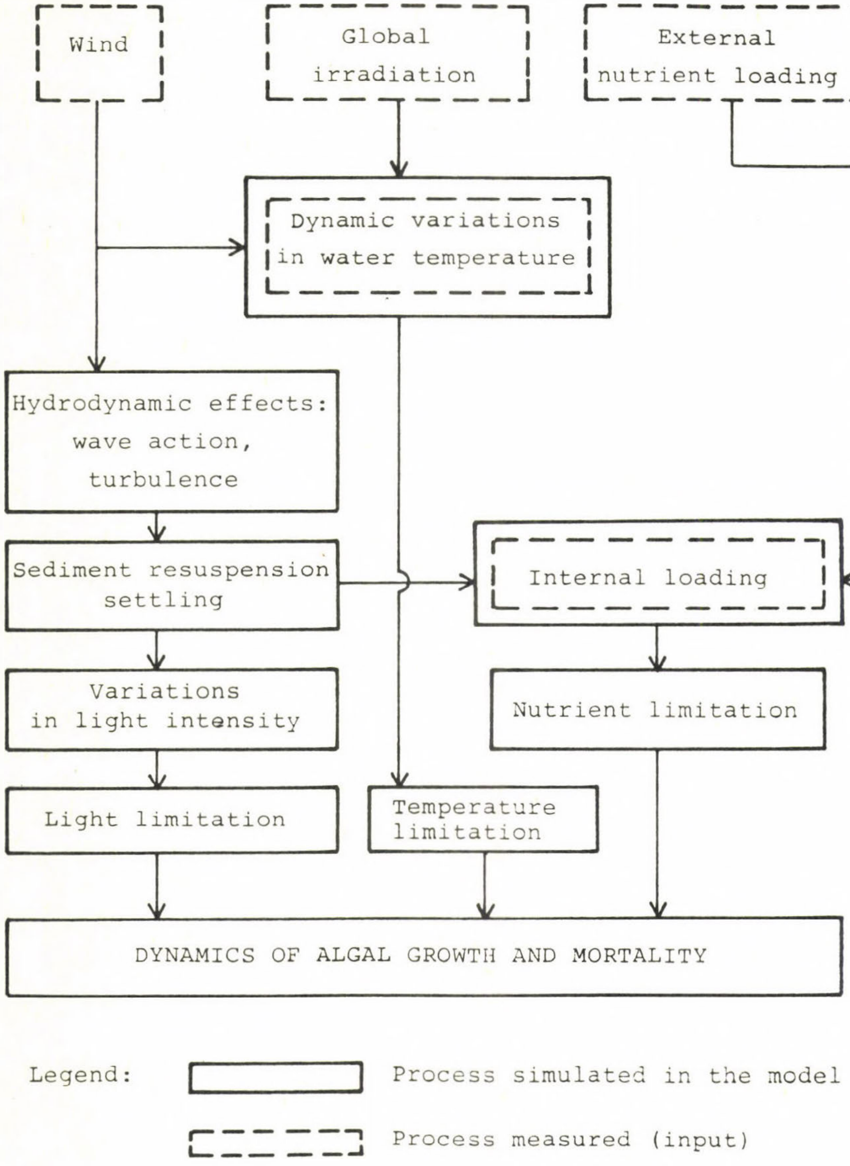


Fig.3. Scheme of the model describing the variations in the phytoplankton biomass

$$FL_i = \frac{1}{T} \int_0^{\tau} \frac{e}{k\,H} \{\exp(\frac{I}{I_{s_i}} \exp(-k\,H)) - \exp(-\frac{I}{I_{s_i}})\}\, d\tau \tag{7}$$

$k = k_o + k_a + k_{ss}$: the extinction coefficient (8)

$k_a = a_{11}\, A_i$: extinction depending on algal self-shading (9)

Extinction coefficient depending on suspended solids:

$$k_{ss} = k_{ss(o)} \exp(-a_3 t) + \frac{a_4}{a_3}\, w\, (1-\exp(-a_3\, t)) \tag{10}$$

Symbols

A_i : biomass of the i-th algal group ($g \cdot m^{-3}$)

D_i^* : maximum mortality rate (d^{-1})

P_i^* : maximum growth rate (d^{-1})

T_{ci}, T_{oi}, β_i: temperature factors on the i-th algal group

N_{oi}, P_{oi}: half-saturation constants

H : water depth (m)

I_{si} : optimal light intensity for the i-th algal group

τ (= 1 day): one-day period

k_o : background extinction coefficient (m^{-1})

k_a : extinction depending on self-shading by the algal biomass (m^{-1})

k_{ss} : extinction depending on suspended solids (m^{-1})

$k_{ss(o)}$: initial (suspended solids dependent) extinction (m^{-1})

a_{11} : self-shading factor

a_3, a_4 : model parameters to be found by calibration (physical background related to resuspension)

w : wind speed (2-hour mean) ($m \cdot s^{-1}$)

P,N : concentrations of nitrogen and phosphorus, respectively, available to the phytoplankton ($g \cdot m^{-3}$)

I : global daily irradiation ($kJ \cdot m^{-2} \cdot d^{-1}$)

T : daily mean water temperature (^{o}C)

t : time (day)

Assumptions underlying the model

(i) The amount of suspended solids transported horizontally is small enough to be neglected relative to that entering the water body by resuspension.
The amount of algae removed by settling of the algal biomass and that transported by horizontal current flow is likewise negligibly small.

(ii) The algal biomass is represented adequately by the volumetric mean by basins.

(iii) The overall variation in algal biomass is described by the variation of the biomass of the four algal groups.
The species with identical population kinetic parameters belong to the same algal group.
In other words this criterion implies that all significant species present belong to one of the algal groups.

The following groups are comprised or simulated in the model:

- The group of winter, spring algae (substantially the diatoms) (a)
- The group of summer algae (b)
- The group of autumn algae (c)
- The nitrogen fixing blue-green algae (d)

(iv) Competition is allowed for indirectly, in that the algal groups interact with each other through light self-shading, while in the competition for nutrients they are favoured or disfavoured by their respective half-saturation constants.

(v) Nutrient availability appears as an input only and is assumed to remain constant over the period simulated. This implies also that the period simulated should be a short one (2-3 weeks).

(vi) The group of blue greens is not subject to light and nitrogen limitation. The light, nutrient and temperature limitations affecting the other groups can be calculated from the foregoing expressions.

(vii) The biomass is calculated by the model (in time increments of 1 day, or two hours for light limitation).

A close correlation is assumed to exist between the biomass and chlorophyll (Németh and Vörös, 1986), so that the biomass can be converted into chlorophyll without any major error.

Input data

For operating the model the following input data are needed:

- Wind speed (2-hour means),
- global irradiation,
- daily mean water temperature,
- the biologically available phosphorus or nitrogen, 2-3 week means,
- water depth,
- the calibrated values (based on in situ measurements) of the parameters a_3, a_4, depending on sediment type,
- the calibrated (species specific) value of the self-shading factor a_{11},
- the kinetic constants calibrated for the four algal groups (Table 2) taken over from the BEM model (Kutas and Herodek, 1983),
- initial values: the biomass of the four algal groups.

In the Keszthely Basin the following values are applicable (VITUKI, 1985):

$a_3 = 1.10 \quad 10^{-4}\ g \cdot m^{-3}$

$a_4 = 1.18 \quad 10^{-4}\ m \cdot s^{-1}$

$a_{11} = 1.90 \quad 10^{-1}\ m^2 \cdot g^{-1}$

Table 2. Kinetic constants for the algal groups a,b,c and d

Constant	a	b	c	d
P^* (d^{-1})	7	3.2	1.1	2
D^* (d^{-1})	3.3	0.85	0.25	0.25
T_c (^oC)	30	21	20	27
T_O (^oC)	23	11.5	6	24
P_O ($g \cdot m^{-3}$)	0.004	0.006	0.009	0.012
N_O ($g \cdot m^{-3}$)	0.04	0.09	0.11	-
I_s ($kJ \cdot m^{-2} d^{-1}$)	5024	3349	1256	-
β ($^oC^{-1}$)	0.065	0.025	0.015	0.2

Analysis

Analysis of the results of limnocorral experiments in 1986 and 1987 has implied that the zooplankton - phytoplankton interaction neglected in the model described may periodically become important enough to be taken into consideration.

The effect of the zooplankton in reducing the phytoplankton biomass is an especially pronounced one in calm weather, at low wind speeds. However, beyond wind speeds of 3 m s^{-1} the mortality of the zooplankton in the Keszthely Basin of Lake Balaton assumes high rates so that its effect becomes negligible. For the wind speed range up to 3 m s^{-1} a significant regression correlation exists between the biomasses of the zooplankton and the phytoplankton.

As an illustration of the foregoing the dynamic model simulation is shown in Fig.4 with and without taking the effect of the zooplankton into account. This effect was approximated by the regression model fitted to the residual term (M) of the simulation.

The residual term M, viz. the difference between the phytoplankton biomass values measured and calculated, is described by the following regression model:

$$M = 10.87 + 0.08\ zoo \quad (r = 0.82) \tag{11}$$

where zoo: the biomass of the zooplankton (mg m^{-3}) and
r : the correlation coefficient.

Verification of the model calibrated for the measurements in 1987 (Fig.4) on the basis of the evaluated measurement data of 1986 is shown in Figs 5 and 6.

It will be perceived that the model does not fit the measurement data as closely as could be expected from the simulation in 1987, nevertheless it reveals adequately the main trends of change.

The differences between the chlorophyll-a values in the open

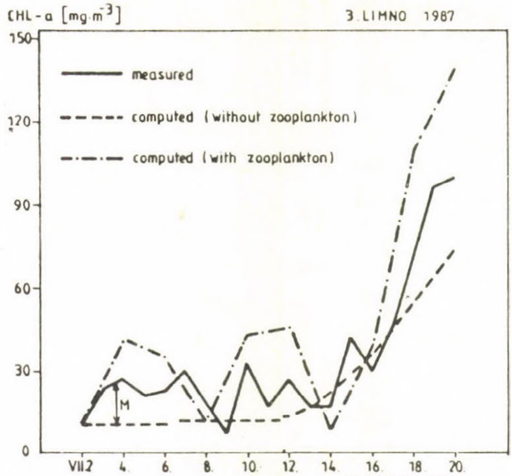


Fig. 4.

Simulated and measured chlorophyll-a concentration in 3. enclosure, 1987

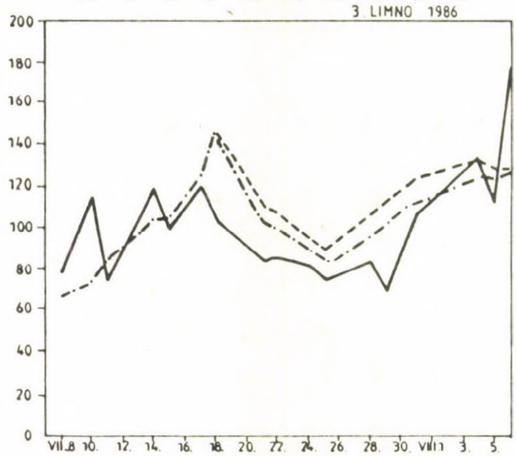


Fig. 5.

Simulated and measured chlorophyll-a concentration in 3. enclosure, 1986

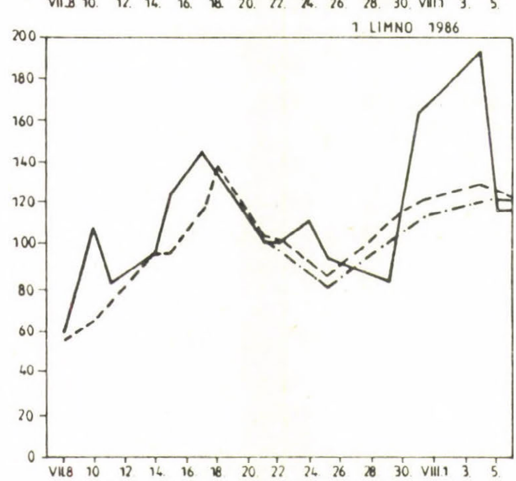


Fig. 6.

Simulated and measured chlorophyll-a concentration in 1. enclosure, 1986

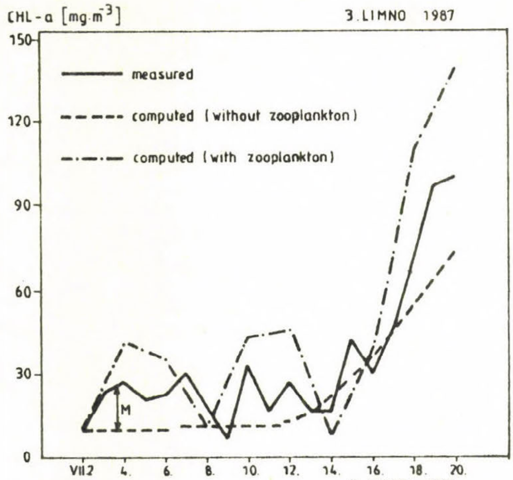


Fig. 7.
Simulated and measured differences in chlorophyll-a concentrations between enclosure no.2. and open water, 1987

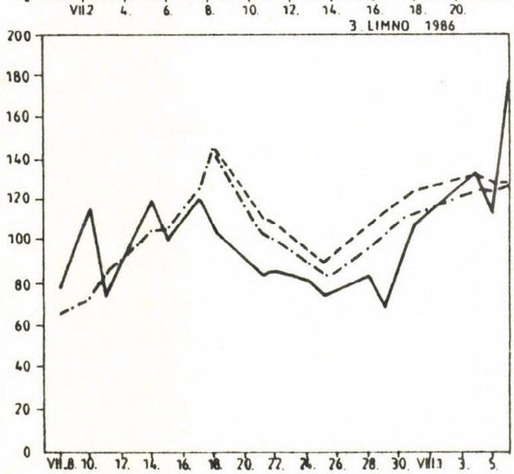


Fig. 8.
Simulated and measured differences in chlorophyll-a concentrations between enclosure no.3. and open water, 1987

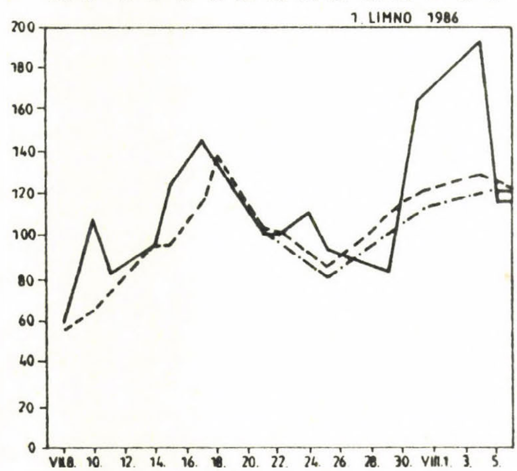


Fig. 9.
Simulated and measured differences in chlorophyll-a concentrations between enclosure no.4 and open water, 1987

lake water and in the enclosures indicate no definite trend, which would reveal any effect of the rate of nutrient loading, although these appear to be attributable to the differences between the zooplankton biomasses measured in the open water and within the enclosures.

By multivariate regression analysis we have succeeded in deriving for the differences between the chlorophyll-a concentrations observed in the open water and in the enclosures the following expression (Figs 7, 8, 9):

$$\Delta\ (\text{chl-a}) = b_o + b_1(\Delta(\text{zoo})) + b_2\ \bar{w} + b_3\ \sigma_w \qquad (12)$$

wherein b_o, b_1, b_2 and b_3 are the regression coefficients, Δ(zoo) is the difference between the zooplankton biomasses in the open water and in the enclosures ($mg \cdot m^{-3}$), $\bar{w}$ is the daily mean wind speed and σ_w the standard deviation thereof.

From Eq.(12) it may be concluded that the presumably hydrodynamic and resuspension differences tend to increase together with wind speed (the sign of b_2 was invariably a positive one for each enclosure), whereas the negative sign of the coefficient b_3, seems to imply that wind speeds of shorter duration produce but minor differences in the phytoplankton biomass. These simple observations appear to lead to the conclusion that the natural hydraulic conditions are modified by the enclosures to an extent depending on wind speed and that the enclosures seem to provide in the range of lower wind speeds a higher degree of sheltering to the zooplankton population, which decreases to a higher extent the phytoplankton biomass under favourable conditions. On the other hand, at higher wind speeds, or at such of longer duration the mortality trend of the zooplankton shows rapidly growing rates tending to balance the conditions in the open water and the enclosure.

The coexistential pattern of planktonic communities

The temporal variations of coexistential pattern was evaluated by correspondence analysis (Legendre, L. and Legendre, P. 1983) with the PRINCOMP programme of the SYN-TAX III. package (Podani 1988). The size of the individual populations are expressed in terms of their biomass (phytoplankton: $mg \cdot m^{-3}$, zooplankton: mg dry weight $\cdot m^{-3}$). The results of analysis have been described in detail in Németh and Vörös (1989).

The temporal variations in the coexistential pattern of phytoplankton was generally similar in all enclosures and in open water, in 1986. The successional sequence of the dominant taxa was as follows:

Anabaena cf. aphanizomenoides —— Anabaena spiroides ——
—— Aphanizomenon flos-aquae (Aphanizomenon issatschenkoi)——
—— Anabaenopsis raciborskii

The filamentous, N_2-fixing blue-green algae were dominant during the whole experimental period. The various treatment caused small but significant changes in the phytoplankton biomass,but did not affect the temporal variation of the coexistential pattern.

The filamentous blue-green algae were present only in a small amount in 1987 at the beginning of the experiment. The phytoplankton were dominated by the taxa belonging to the Euglenophyta, Cryptophyceae and the diatoms (Centrales). The algal bloom caused mainly by Anabaena spiroides started only 10th of July in the open water.

Enclosure no.1

experiment no.1: Euglena spp. —— Rhodomonas minuta ——
—— Cryptomonas spp. (Euglena spp.)

experiment no.2: Filamentous heterocystous blue green algae and Gomphosphaeria lacustris; no definite trend could be recognized in relative abundances of dominant taxa

Enclosures no. 2, 3 and 4

experiment no.1.: Cyclotella spp. (Euglenophyta spp.) ——

—— Cryptomonas spp. (Melosira granulata var. angustissima and/or Pennales spp.)

experiment no.2: filamentous heterocystous blue-green algae (Anabaena spiroides, Anabaenopsis raciborskii) —— Gomphosphaeria lacustris (Cryptomonas spp.)

Lake Balaton (no.5)

Anabaena cf. aphanizomenoides (Cyclotella spp., Stephanodiscus spp., Euglena spp.) —— Anabaena spiroides (water bloom) —— Aphanizomenon flos-aquae (Anabaenopsis raciborskii, Aphanizomenon issatschenkoi) —— Gomphosphaeria lacustris (Cryptomonas spp.).

The temporal variations in the coexistential pattern of the phytoplankton remained thus (in agreement with the result of 1986) unaffected by nutrient loading of different rates and N/P ratios.

However, under the influence of the sediment dredged from 4-5 m depth and spread on the surface of the bed sediment (cover on the layer containing spores of blue-greens, higher ammonia concentration, altered light conditions of a few days duration), the phytoplankton structure and the temporal variations therein were also modified relative to the other enclosures and to the open water of Lake Balaton.

Zooplankton

The following successional sequences in the enclosures and in the open water were observed in 1986:

Enclosure no.1: Diaphanosoma brachyurum (DIA) —— (Nauplius larvae (N), Daphnia cucullata (DC), Eudiaptomus gracilis (EG), Daphnia galeata (DG)) —— copepodite larvae male (K♂), Acanthocyclops robustus (AR) —— copepodite larvae (K)

Enclosure no.2: DIA —— (N, DC, DG, EG) —— K♂ , AR —— K

Enclosure no.3: DIA —— (N, DC, DG, EG) —— AR, K, K♂

Enclosure no.4: DIA —— (N, DC, DG) —— (EG) —— AR, K, K♂

Lake Balaton (no.5): DIA —— (DG, DC, N) —— AR, (K) —— ML (K♂)
(ML: Mesocyclops leuckarti)

The variations in the structure of the filter-feeding zooplankton were substantially identical in each enclosure, as anticipated.

The correspondence analysis has lead to the following simplified description of the temporal variations in the coexistential pattern of the zooplankton in 1987:

Enclosure no.1

experiment no.1: (Eudiaptomus gracilis (EG) —— Diaphanosoma brachyurum (DIA), copepodite larvae 1 (K1) —— Daphnia cucullata (DC), Daphnia galeata (DG) —— Acanthocyclops robustus (AR), copepodite larvae 2 (K2), copepodite larvae male (K ♂)

experiment no.2: no definite trend in the temporal variations, the zooplankton was dominated by K2, K♂ and AR

Enclosures no.2, 3 and 4

experiment no.1: EG —— DIA —— DC, DG —— K1, K2, K ♂ , AR

experiment no.2: no definite trend in the temporal variations, the zooplankton was dominated by K1, K2, K♂ , AR

Lake Balaton (no. 5): No definite trend could be recognized in the coexistential pattern of the zooplankton in the open water of Lake Balaton over the period studied. The dominant taxa were Diaphanosoma brachyurum in the period from the 2nd to the 22nd July, 1987 and normally copepodite larvae or Acanthocyclops robustus between the 24th July and 7th August, 1987.

CONCLUDING REMARKS

The set up of the enclosure experiment has assured the evaluation the sensitivity of the model structure to the randomly variable meteorological factors (global radiation, wind speed and water temperature) as well as the filter-feeding zooplankton mainly because of the system was not nutrient limited by using appropriate nutrient loading.

It is concluded that in that area of Lake Balaton the phytoplankton biomass and community structure is highly regulated by the filter-feeding zooplankton.

The similarity in the temporal variations of community structure which has been found regardless to the different nutrient loadings supports the hypothesis,that in this hypertrophic region of the lake,the aquatic ecosystem is not nutrient limited.

Substantial effect had however observed by adding sediments originated from the deeper layer on the temporal variations in the coexistential pattern of planctonic communities.

REFERENCES

Bottrell,H.H.,Duncan,A.,Gliwicz, Z.M.,Grygierek,E., Herzig,A., Hillbricht-Ilkowska, A., Kurasawa, H., Larsson, P. and Weglenska, T. 1976: A review of some problems in zooplankton production studies. Norw. J. Zool. 24, 419-456.

Felföldy L. 1987: A biológiai vizminősités. Vizügyi Hidrobiológia 16, 1-258

Istvánovics, V., Vörös, L., Herodek,S., G.Tóth,L. and Tátrai,I. 1986: Changes of phosphorus and nitrogen concentration and phytoplankton in enriched lake enclosures. Limnol.Oceanogr. 31(4), 798-811

Kutas,T. and Herodek,S. 1983: BEM, a complex model for simulation Lake Balaton ecosystem. Eutrophication of shallow lakes: modelling and management.IIAS collaborative series.

Leglendre,L. and Legendre,P. 1983: Numerical ecology. 1-419. Elsevier Sci. Publ.Co., Amsterdam, Oxford, New York

Németh J. és Vörös L. 1986: Koncepció és módszertan felszini vizek algológiai monitoringjához. OKTH Környezet- és természetvédelmi kutatások 5, 1-136.

Németh,J. and Vörös,L. 1989: Temporal variations in coexistential pattern of phytoplankton in enriched lake enclosures. Abstracta Botanica (in press).

Podani,J. 1988: SYN-TAX III. User's Manual Abstracta Botanica 12, Suppl. 1, 1-183.

VITUKI,1985: A felkeveredés hatása a Keszthelyi-öböl fényviszonyaira. Témajelentés, 7614/3/36

Symp. Biol. Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

HYDROOPTICAL MODEL TO INVESTIGATE SOME WATER QUALITY COMPONENTS

A.A. GITELSON, F. SZILÁGYI,* G.P. GARBUZOV*

Institute for Hydrochemistry
344090 Rostov-on-Don, Stach'ki Av. 198, USSR
*Research Centre for Water Resources Development
H-1453 Budapest, Pf. 27, Hungary

Abstract

New possibilities for monitoring the quality of surface waters have been established during the last two decades by launching satellites that could fulfill such remote sensing tasks. In the field of water quality monitoring space photography could only compete with the traditional taking of point samples when photo-interpretation can be made without necessitating simultaneous reference samples on the Earth's surface. The first step along this line is to find optically characteristic water quality components and relate them to the spectral characteristic of light reflected from the water surface (i.e. the development of a hydrooptical model). By connecting such hydrooptical models to atmospheric correction models space photographs become interpretable even in the lack of Earth based reference data. The study describes a simple regression-type hydrooptical model that has been developed in the framework of a joint Soviet-Hungarian programme on remote sensing. The model proved to be applicable for the simultaneous interpretation of space photographs for the suspended solids, chlorophyl-a and dissolved organic carbon components of water. In the chlorophyl-a concentration range of 1-150 mg/m^3 the highest standard error of model estimation was 3 mg/m^3, while that of the suspended solids was 5 g/m^3, in the concentration range of 5-40 g/m^3. The assumption of constant model parameters has been checked for Soviet (Azovian sea, River Don and Northern Doniec) and Hungarian (Lake Balaton) test areas.

1. Introduction

The quantity of pollutants discharged into surface waters has significantly increased during the past decades. The earliest water quality problem (in the sixties) associated with the increased loads of plant nutrients was the eutrophication of lakes and reservoirs. Micropollutants and acidic depositions showed their adverse effects in the seventies and eighties. With the recognition of unfavourable water quality tendencies regular water quality monitoring became increasingly important.

The quality of surface waters changes significantly both in time and space. The sampling strategy (the number of sampling points and components and the frequency of sampling) shall be such as to allow the representative qualitative characterisation of the water body in concern. Water quality monitoring has, since many decades, relied mostly on the collection of point samples and their laboratory analysis. The disadvantage of point sampling is that the information assured by it relates only to the immediate vicinity of this point, while the densening of sampling networks has strong economic limitations. Along with the technical development of monitoring instruments continuous sampling, either in time or in space, has - in respect to some quality constituents - became recently possible (Hoffmann et al. 1985). By establishing an appropriately dense monitoring network continuous measurements can provide sufficiently representative time series to be used for the spatial evaluation of water quality changes. Water quality mapping, however, will not be adequately supported even by these methods, since sampling networks of large water surface cannot be made sufficiently dense at reasonable expenditures.

With the increase of the size of water surfaces the monitoring of water quality of the conventional sampling technique type becomes either too costly and time consuming, or results in significant losses of information. Thus the cost effectiveness of these traditional methods cannot be assured in most of the cases. Owing to these facts research into the use of remote sensing methods in the field of water quality monitoring has been given a new impetus.

2. Monitoring of water quality by remote sensing techniques

In the field of water quality monitoring remote sensing methods utilize the information contained in the spectral properties of light radiated onto water surfaces and reflected by them; as these properties change also in function of the mass and type of substances contained in the water. Remote sensing methods of water quality monitoring are, when applied at lower altitudes (from boats or helicopters), very similar to those of the continuous field measurements. When applied at high altitudes (from airplanes or satellites) they can provide instantaneous map-like information over large water surfaces.To make use of such information, however, appropriate computation tools (algorithms) must be made available.

In the earlier phases of using remote sensing methods for water quality monitoring imagery provided by satellites (for example LANDSAT) that had been launched for purposes other than quality monitoring, were utilized. Relationships between earth reference data and the intensity data of space imagery were sought (Ritchie et al. 1976, Smith and Addington 1978, Lindell 1981, Alföldi and Munday 1978).

It was demonstrated by research results that simultaneous Earth reference data show significant correlation with light intensities measured by the satellites in certain spectral ranges for components such as suspended solid concentration (Ritchie et al., 1976, Szabó and Szilágyi 1983), chlorophyl-a concentration (Lindell 1981, Büttner and Vörös 1980, Szabó and Szilágyi 1983) and dissolved organic matter (Szabó and Szilágyi 1983). Similar relationships have been found between the data of the Coastal Zone Color Scanner (CZCS) of satellite "Nimbus" and the chlorophyl-a concentration of water (Johnson 1978, Pan et al. 1988, Mortimer 1988). This scanner was launched for monitoring coastal marine water quality, worked in small bands and had thus low resolution capabilities.

Nevertheless relationships elaborated between the intensity data of satellite sensors and the Earth reference data could not be adapted to space images taken at times other than that of the calibration. The likely explanation of this situation is that atmospheric conditions were different. Since about 80 % of the light measured by the satellite sensors originates from the atmosphere there is a strong need for the development of atmospheric correction models to be associated with the above regression models. When such correction models are not available most of the advantages of space imagery will be lost (i.e. there is a real advantage only when no simultaneous Earth reference data are needed). All these make the use of space imagery in the field of water quality monitoring a fairly difficult task, for which the following problems shall be solved;

i) Relationships between the characteristics of light reflected from water surfaces and the hydrobiological properties of water shall be elaborated along with algorithms of interpretation that make the evaluation independent of the conditions of measurement (i.e. the elaboration of the hydrooptical model of surface waters is needed);
ii) The determination of relationships between radiation energy at the water surface and that measurable at differing altitudes of the atmosphere shall be carried out (i.e. the development of an atmospheric model and a radiation model);
iii) Making use of the above two results an interpretation algorithm of space imagery data shall be developed, that can enable the use of remote sensing data for water quality monitoring without requiring simultaneous Earth reference data. This task means the coupling of models to be developed under items i) and ii).

In our past research work the main emphasis has been laid on the development of hydrooptical models. In respect to atmospheric correction models the following works can be referred to: Munday (1983), Guan et al. (1985), Gordon (1978), and Deschamps et al. (1983).

3. Test areas and methods of measurements

For the development of hydrooptical models a bilateral Soviet-Hungarian scientific co-operation has been established in 1981. The institutions taking part in this co-operation are: Hydrochemical Institute of Rostov (GHI) (from the Soviet part) and Research Centre for Water Resources Development (VITUKI), Budapest (from the Hungarian part). The test areas were as follows: River Don (1983, 1984); the Northern Doniec river (1983, 1984), the Azovian Sea (1984) and Lake Balaton (1985, 1986). In terms of water quality characteristics the test areas are of much differing type.

River Don is characterized primarily by high concentrations of clayey inorganic suspended solids, while the Northern Doniec river by those of dissolved and particulated organic matter. Chlorophyl-a concentrations are relatively low, below 50 mg/m^3, in both rivers. The water of Azovian Sea is oligotrophic with low dissolved organic matter and suspended solid content. The water of Lake Balaton, Hungary, belongs to the eutrophic-hypertrophic category. Due to the shallowness of this lake (average depth; 3.14 m) resuspension caused by wind-induced wave action is basically affecting the suspended solid content of water (varying between 1 and 250 g/m^3). The dissolved organic matter concentration of the water of Lake Balaton is low (1-10 g/m^3).

The energy of light radiated onto and reflected from the water surface was measured by instrument "Spektrum -ol" that had been developed in institute GHI. With this instrument the energy of light can be measured and registered on a continuous basis for each wave length in 430-750 nm interval. The instrument is based on a prismatic monochromator, while scanning is made mechanically. The instrument was mounted either on a boat or on a helicopter. The altitude of measurements varied between 2.0 and 100 meters, depending on the type of the application. During the past period of the investigations several hundred spectral surveys have been carried out.

Simultaneously with the spectral measurements Earth reference data have been also collected. The water samples were analyzed for chlorophyl-a and suspended solid content, using the methods proposed by Felföldy (1980) and the UMWA (1973). The fluorescence of fulvic and humic acids, being proportional to their concentrations, was measured by a Soviet made instrument "Kvant-5" spectrofluorimeter.

4. Hydrooptical model of surface waters

From the viewpoint of hydrooptics surface waters can be considered as two-phase systems (dissolved and suspended phases) in which both phases are of the multiple-component type. Some of these components will affect the light reflected back from the water surface through their light-scattering, absorbing or emitting properties, while others will not have such effects. Components having special spectral properties can be subjected to remote sensing, while the others cannot. In the first step some of the general water quality components shall be selected for remote sensing investigations. Of these the most important ones are: the concentrations of inorganic suspended solids (C_{SS}), chlorophyl-a (C_{CHL}) and dissolved organic carbon (C_{DOC}).

The general quality characteristics of the water body (for example C_{CHL}, C_{SS}, C_{DOC}) cannot be directly determined on the basis of the spectrum and intensity of light reflected from the water surface, since the relationship between these variables is not straightforward. The water quality information contained in the properties of the reflected light is, due to complicated spectral interactions, of hidden character. Nevertheless, it can be assumed at the same time that all spectral properties will represent a certain combination of water quality parameter values, which latter will determine, at the given wavelength or band, the optical properties of water. Figure 1 shows the radiation energy of light radiated onto and reflected from the water surface, for mesotrophic water bodies and in the wavelength domain of visible light. Although these spectra carry water quality information, they cannot be directly

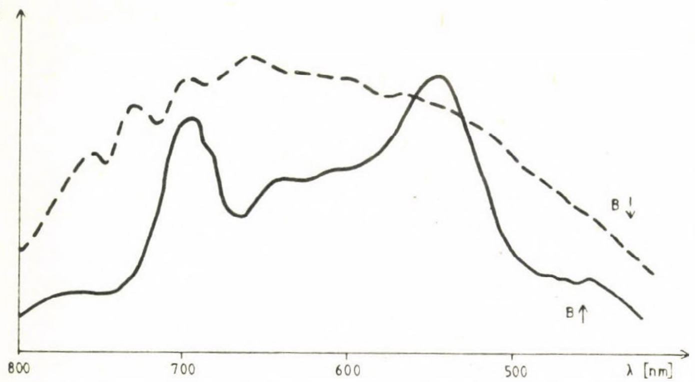


Figure 1. Energy of light (E) radiated onto (B↓) and reflected from (B↑) the water surface as a function of the wavelength (λ); (Lake Balaton,Szigligeti Basin) (mark B↑ is amplified 3-fold)

used for drawing conclusions on the state of water quality.The decoding of water quality information can be made as follows:

i) The energy of light ($B\uparrow(\lambda)$) scattered back from the water surface shall be made independent of that radiated onto the water surface ($B\downarrow(\lambda)$) and shall be related to the energy of light scattered back from an ideally reflecting surface at each of the wavelengths ($B_o\uparrow(\lambda)$). This means that the spectral radiation coefficients ($\rho(\lambda)$) shall be calculated as:

$$\rho(\lambda) = \frac{B\uparrow(\lambda) - 0.02\ B\downarrow(\lambda)}{B_o\uparrow(\lambda)} \qquad (1)$$

The value of $\rho(\lambda)$ as obtained from Equation 1 will not depend on the energy of light radiated onto the water surface. The course of this curve will characterize the composition of water. $\rho(\lambda)$ functions obtained for three areas of Lake Balaton, having differing water quality characteristics, are shown in Figure 2.

ii) As indicated by Figure 2, the shapes of $\rho(\lambda)$ curves are similar to each other, and their differences can be related to the differences in the water quality properties of the respective water bodies. Since values ρ_{max} are different, the rela-

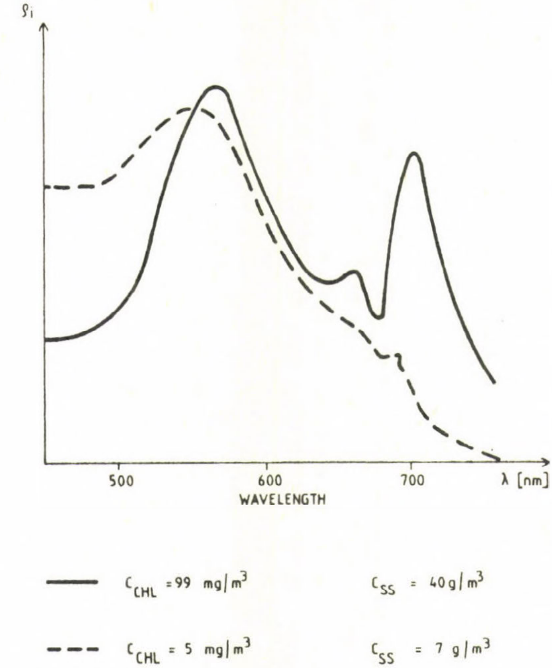


Figure 2. Relationship $\rho(\lambda)$ for the Lake Balaton test area (1985)

tionship must be normalized. Curves normalized for value ρ_{max} are shown in Figure 3.

iii) In the next step wavelengths enabling the most sensitive measurement of the selected water quality components shall be identified. When selecting the appropriate wavelength it shall be assured, in addition to assuring appropriate measuring sensitivity, that the determination of one of the components is not to be disturbed by the presence of another one. This can be solved by applying factor analysis (Gitelson et al. 1985), although some other researchers use the method of stepwise regression analysis (Johnson, 1978). Both of these methods are equally suitable for selecting the wavelengths (domains) for measuring the desired quality components. The result of factor analysis made for the data measured in Lake Balaton in 1985 is shown in Figure 4. The figure indicates the fraction of standard deviation, considered a unit, that can be explained by the factor in concern.

The weighting coefficients of one of the factors are the greatest in the range of 700-718 nm. Based on control measurements with algae cultures it was found that this factor can be considered as that representing the state of phytoplankton populations. Consequently relationship C_{CHL} (ρ_{700}) can be considered as the one most suitable for the remote sensing measurements of chlorophyl-a concentrations. The factor of maximum

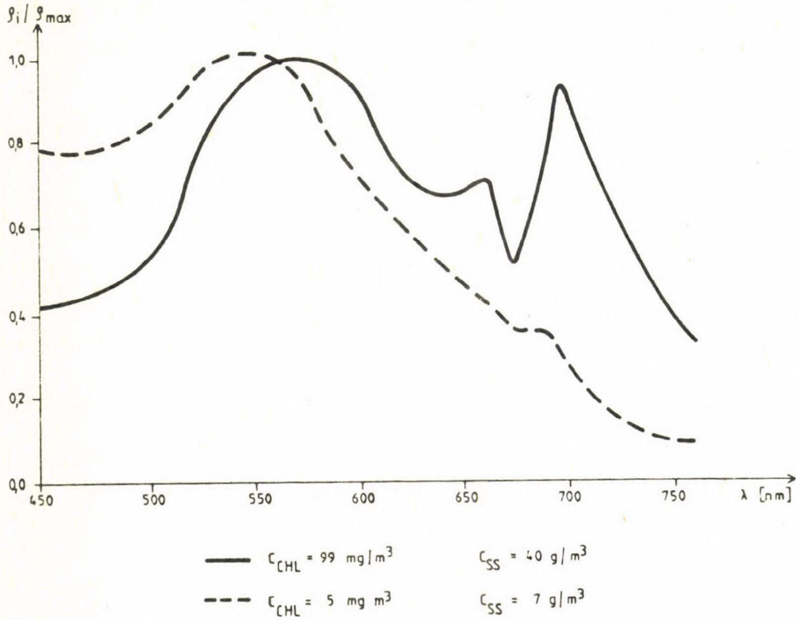


Figure 3. Relationship $\rho_i / \rho_{max} (\lambda)$ for the Lake Balaton test area (1985)

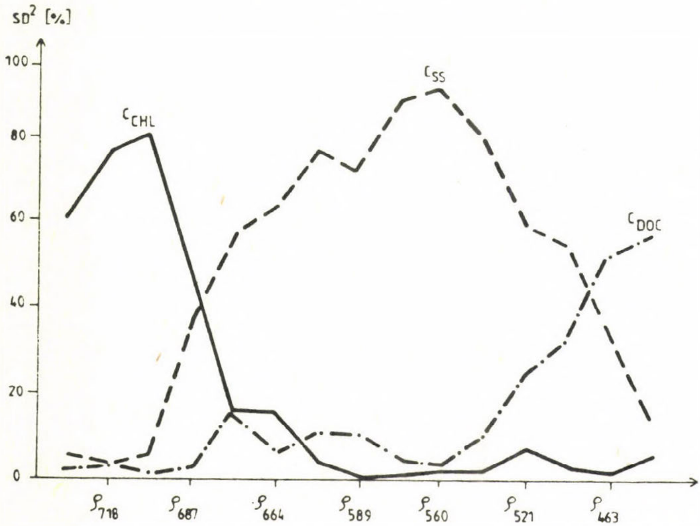


Figure 4. Results of factor analysis based on the non-standardized data of Lake Balaton measurements. Numbers assigned to the curves mean the fraction of the standard deviation that is explained by the respective factor (the residual standard deviation is 15.4 %). SD^2 = square of standard deviation

weighting coefficient in band 560-570 nm will, most likely, represent the light scattered by the suspended solid particles. Namely, it is well known that organic and inorganic suspended solid particles will exercise maximum light scattering in the vicinity of wavelength 550 nm.

The third factor found at around 460 nm is strongly related to dissolved (particulated plus dissolved) organic matter that absorbs light effectively in the blue domain of the spectrum. At a given wavelength, however, the spectral characteristics of radiation will not be affected by a single component only. Consequently a factor cannot be considered as exclusively identical with one or another one of the components. Thus, on the basis of the results of factor analysis wavelength ranges 700-720 nm, 520-560 nm and 450-470 nm seem to be the most suitable for remotely sensing the concentrations of C_{CHL}, C_{SS} and C_{DOC}, respectively. In developing the models several $\rho(\lambda)$ vs. concentration functions have been tested for C_{CHL}, C_{SS} and C_{DOC}. In this study the following relationships expected to yield the best results, will be evaluated:

a) ρ_{700}/ρ_{560} (C_{CHL})

b) $\dfrac{\rho_{560}-\rho_{520}}{\rho_{560}+\rho_{520}}$ (C_{SS})

c) $\dfrac{\rho_{430}-\rho_{620}}{\rho_{430}+\rho_{620}}$ (C_{DOC})

In evaluating these relationships the main emphasis has been laid on the results of remote sensing C_{CHL}, since this component is one of the most important measures of the trophic state of waters.

Figure 5 shows the results achieved by the investigations made on the Soviet test area in 1983 and 1984. The ranges of C_{CHL} and C_{SS} concentration values were 1-40 mg/m^3 and 5-40 g/m^3, respectively. In spite of the fact that measurements were made in much differing water bodies and in different times the ob-

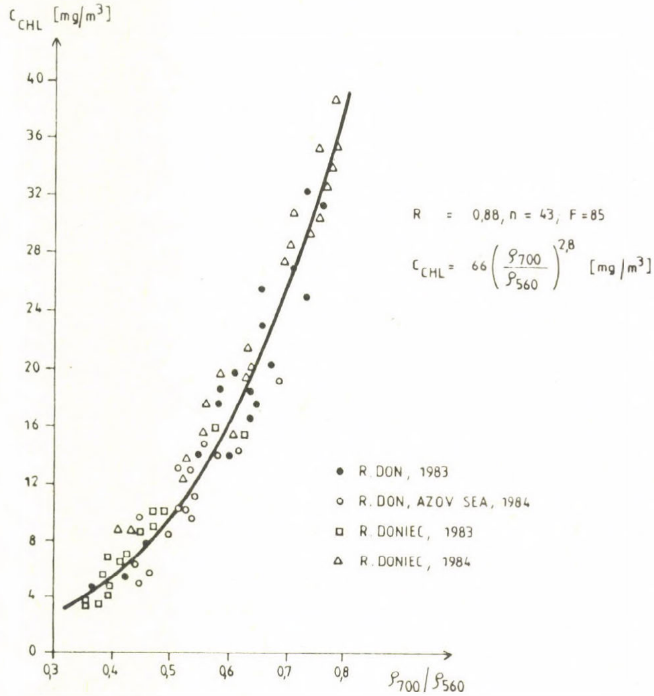


Figure 5. Relationship ρ_{700}/ρ_{600} (C_{CHL}) for the Soviet test area
R = coefficient of correlation

tained ρ_{700}/ρ_{600} (C_{CHL}) relationship was sufficiently close, both in terms of the correlation coefficient and the value of factor F.

During the Lake Balaton measurements in 1985 the concentration ranges of C_{CHL} and C_{SS} were 5-150 mg/m^3 and 5-40 g/m^3, respectively.

Evaluating the results of Lake Balaton measurements, made in two different times in 1985, together (Figure 6) it was found that the relationship between ρ_{700}/ρ_{560} and C_{CHL} was also very close. As indicated by the results of error analysis the lowest standard error of C_{CHL} estimation made on the basis of ρ_{700}/ρ_{560}, was 2.8 mg/m^3. Considering the wide C_{CHL} ranges of the measured values this standard error can be considered a very favourable one.

Comparing the parameters of the relationships obtained for the Soviet and Hungarian test areas, respectively, it was found that the coefficient of the Lake Balaton relationship was twice as high as that of the Soviet test area, while the exponent of the former was 20 % smaller than that of the latter (Figures 5 and 6). The likely cause of this discrepancy is the differing concentration domain of chlorophyl-a, in the two test areas. Further investigations would be needed to find answers to the following questions:

i, are the parameters of the two relationships significantly different from each other?, and

ii, are the parameters of a relationship obtained for a given water body constant, or vary in time?

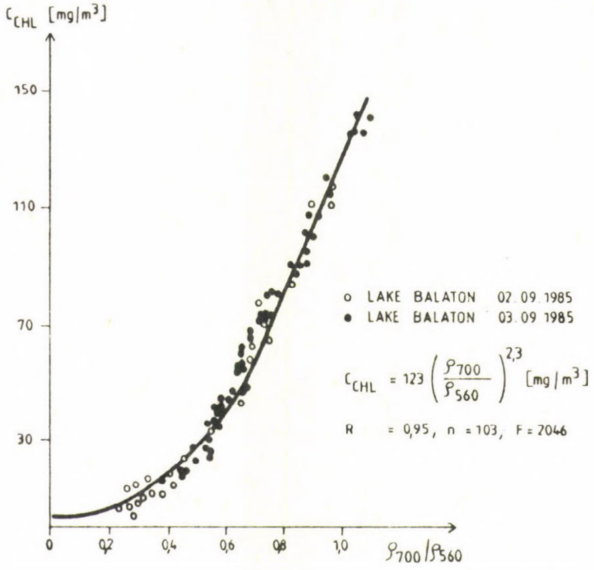


Figure 6. Relationship $\rho_{700}/\rho_{600}/C_{CHL}$ for the Lake Balaton test area
R = coefficient of correlation

For combination $\rho_{560}-\rho_{520}/\rho_{560}+\rho_{520}$, applied for describing C_{SS}, the data of the Soviet test areas (River Don and the Azovian Sea) yielded the following expression:

$$C_{SS}(g/m\ ) = 61\left(\frac{\rho_{560} - \rho_{520}}{\rho_{560} + \rho_{620}}\right)^{0.49}$$

$$R = 0.93;\ n = 63;\ F = 405$$

In the C_{SS} concentration range of 5-50 g/m³ the standard error of the estimation was 3.2 g/m³. The processing of C_{SS} data for the Lake Balaton study is under way.

The development of decoding feature for C_{DOC} estimation is complicated by the fact that C_{DOC} is closely related to C_{CHL} for most of the water bodies studied. Thus for Lake Balaton and the Azovian Sea C_{DOC} was estimated from the decoding features developed for C_{CHL}. For example in the case of Lake Balaton C_{CHL} is closely related to C_{DOC} (R 0.9; n=54), and thus the $\rho_{700}/\rho_{560}(C_{DOC})$ relationship allows the estimation of C_{DOC} with an error less than 0.5 gC/m^3.

If there is no correlation between C_{CHL} and C_{DOC} $\rho_{430}-\rho_{620}/ /\rho_{430}+\rho_{620}$ seems to be the best combination of $\rho(\lambda)$ for estimating C_{DOC}. For River Northern Doniec this relationship, together with the model, allowed the estimation of C_{DOC} with an error less than 0.65 gC/litre.

Of the models described above the time dependency of the parameters of the Lake Balaton relationship $\rho_{700}/\rho_{560}(C_{CHL})$ has been already investigated. A function similar to that developed with the data of 1985 (Figure 6) has been fitted to the data of 1986. The relationship thus obtained is as follows:

$$C_{CHL}\,(mg/m^3) = 140\left(\frac{\rho\,700}{\rho\,560}\right)^{2.7}$$

R = 0.94; n = 80; F = 2700

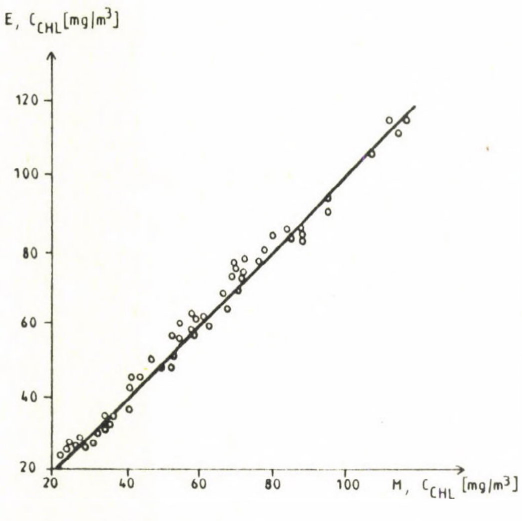


Figure 7. Chlorophyl-a concentration (C_{CHL}) in Lake Balaton as measured (M) in 1986 and estimated by the hydrooptical model (E) made in 1985.

It can be stated that the parameters of the models calibrated against the data of 1985 and 1986 respectively are similar to each other. Figure 7 shows the C_{CHL} values of 1986, as estimated with the model developed for the data of 1985, in function of the measured laboratory C_{CHL} data of 1986. It can be seen that the measured and estimated values are closely fitting. The average squared error between estimated and measured values was smaller than 2.3 mg/m^3. It can be thus assumed that for C_{CHL} the parameters of the Lake Balaton relationship are constant in the summer period. Nevertheless further measurements would be needed for the full verification of this assumption.

References

Alföldi, T.T. and Munday, J.C. (1978): Water quality analysis by digital chromaticity mapping of Landsat data. Can. J. of Remote Sensing 4(2): 1291-1298

Büttner, Gy. and Vörös, L. (1980): Investigation of Hungarian lakes using Landsat MSS data. MTESZ,Budapest, 1980 (in Hungarian)

Deschamps, P.Y., Herman, M. and Tanre, D. (1983): Modeling of the atmospheric effects and its application to the remote sensing of ocean color. Applied Optics 22(34): 3751-3758

Felföldy, L. (1980): Biological water quality classification. Vizügyi Hidrobiológia 9. VIZDOK, Budapest, pp. 266. (in Hungarian)

Gitelson, A.A., Nyikanorov, A.H., Szabó Gy. and Szilágyi, F. (1985): Etude de la qualité des eaux de surface par télédetection. In: D. Lerner (ed.): Monitoring to Detect Changes in Water Quality Series, IAHS No. 157. 111-122

Gordon,H.R. (1978): Removal of atmospheric effects from satellite imagery of the oceans. Applied Optics 17(10):1631-1636

Guan, F., Pelácz, J. and Stewart, R.H. (1985): The atmospheric correction and measurement of chlorophyll concentration using the coastal zone color scanner. Limnol. Oceanogr.30(2): 273-275.

Hoffmann, I., Szilágyi, F. and Gitelson, A.A. (1984): Application of automatic mobile systems in water quality investigations. Vizügyi Közlemények 66: 126-139. (in Hungarian)

Johnson, R.E. (1978): Mapping of chlorophyl-a distribution in coastal zones. Photogrammetric Engineering and Remote Sensing 44(5): 617-624

Lindell, L.F. (1981): Mapping of water quality using Landsat Imagery. Proc. 15th Int. Symp. on Remote Sensing of Environment, Ann Arbor, Michigan

Mortimer, C.H. (1988): Discoveries and testable hypotheses arising from Coastal Zone Color Scanner imagery of southern Lake Michigan. Limnol. Oceanogr. 33(2): 203-226

Munday, J.C. (1983): Chromaticity of path radiance and atmospheric correction of Landsat data. Remote Sensing of Environ. 13: 525-538.

Pan, D., Gower, J.F.R. and Borstad, G.A. (1988): Seasonal variation of the surface chlorophyl distribution along the British Columbia coast as shown by CZCS satellite imagery. Limnol. Oceanogr. 33(2): 227-244.

Ritchie, J.C., Schiete,F.R. and McHenry, J.M. (1976): Remote sensing of suspended sediments in surface waters. Photogram. Eng. Remote Sensing lq(12): 1539-1545.

Smith, A.Y. and Addington, J.D. (1978): Water quality monitoring of Lake Mead: A practical look at the difficulties encountered in the application of remotely sensed data to analysis of temporal change (manuscript). Presented to the 5th Canadian Symposium on Remote Sensing, Victoria, August 1978

Szabó, Gy., Szilágyi, F. (1983): The comparison of Landsat images used for water quality investigation at Lake Balaton, Hungary. Vizügyi Közlemények 65: 401-416. (in Hungarian)

UMWA (1973): Unified methods of water analysis. Himiya, 1973, p. 47-49 (in Russian)

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

POND RESTORATION BY USING ALUM TREATMENT AND SEDIMENT REMOVAL IN NE HUNGARY

Gy. LAKATOS, J. SOÓS

Institute of Ecology, Kossuth Lajos University
H-4010 Debrecen, Hungary

INTRODUCTION

The development of modern society has had great impacts on many of the world's water bodies (Forsberg 1987). The effect of nutrient enrichment on shallow ponds can be observed all over the world, and phosphorus has been implicated as a main factor stimulating unwanted processes of eutrophication (Gächter 1987, Schindler 1985, Vollenweider 1968).

Shallow ponds play a significant role in the water management of Hungary, their biological water quality, however, is rapidly deteriorating - especially in areas with great anthropogenic influences (Dévai et al. 1979).

This paradoxical situation has become obvious in the case of Pond Sóstó (PS) close to the town Nyíregyháza in NE Hungary. This pond was a popular resort place for the population of this region until the sixties. From that time on the water quality has undergone drastic changes due to the rapid cultural and planktonic eutrophication (Dévai et al. 1979). As a consequence, not only bathing has become impossible but the pond's aesthetic value also has decreased.

There were two objectives in our study on this pond. First, we carried out chemical and biological analyses of the water bodies in order to establish the pond parameters expected to be responsible for the development of the bad water quality. Second, taking into consideration the findings of previous studies (Dévai et al. 1979, Kollár et al. 1978, Lakatos 1978), we applied effective and economic restoration methods.

Restoration of water bodies involves a set of treatment modes which are suitable for achieving their homeostatic response to a human impact and improving their water quality. The perturbation calling for restorative measures may result from natural as well as human processes and they affect pond characteristics such as water budgets and water quality (Lerman and Hull 1987). The concept of lake restoration measures was developed by Björk (1985) in the mid-1960s.

In this paper we review the hydrobiological responses of PS to the effects of restoration based on water exchange. Furthermore, we are going to summarize the chemical and biological effects of internal phosphorus and nitrogen loading which has been reduced by the application of aluminium sulphate and sediment removal.

MATERIAL AND METHODS

The scheme of PS and the sampling points are demonstrated in Fig. 1. At present, the pond consists of two different parts, the Bathing pond (B) and the Rowing pond (R). These ponds are totally closed without any in- and outflow, so their water level is a function of melting, precipitation and ground water. The catchment area of the ponds is relatively small (about 9 hectares) and it is affected by intensive park management (Table 1).

Table 1. Important data of hydromorphometry of the pond in 1974 and later of the Bathing (B) and the Rowing pond (R)

	1974	1984		1986	
		B	R	B	R
Area (ha)	9.5	4.4	5.1	4.4	5.1
Volume ($10^3 \cdot m^3$)	152	88	107	80	92
Mean depth (m)	1.6	2.0	2.1	2.0	1.8
Maximum depth (m)	1.8	2.1	2.3	2.1	2.2
Transparency (cm)	15	50	40	35	31
Shoreline (m)	1646	872	974	870	970

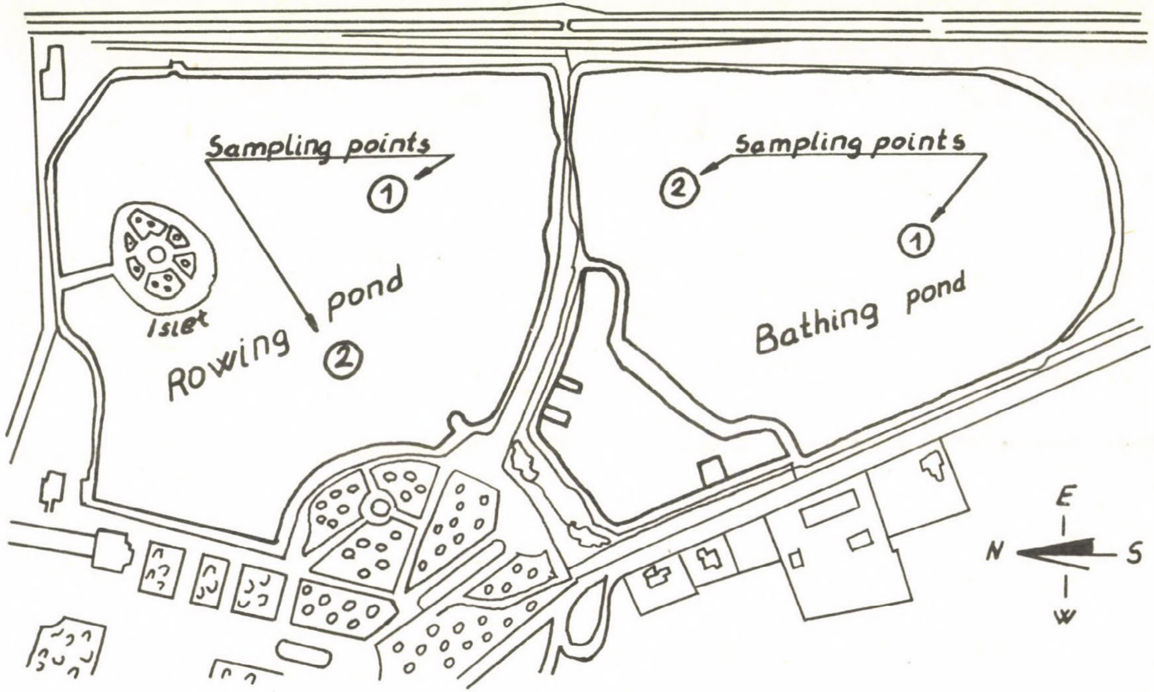


Fig. 1. The outline of Pond Sóstó and the sampling points

The original PS had an area of 9.5 hectares and it was very shallow with a slight transparency. In 1975 a wide dam was constructed to separate the Bathing and Rowing ponds which presently have almost the same areas and volumes. The new deeper water bodies have higher transparency values than the original pond. It seemed evident from our earlier hydrobiological findings (Dévai et al. 1978, Kollár et al. 1978, Lakatos 1978) that the unfavourable conditions were due to the planktonic eutrophication of these water bodies. This can be changed by the introduction of restoration methods. Naturally, we have searched the ponds and have performed many laboratory and field studies (Dévai et al. 1979), in this paper, however, we are not going to deal with their results.

In 1975, during the separation of the new ponds, water exchange was performed. In the summer of 1984 alum of $50\,g \cdot m^{-3}$ was added to the Bathing pond in contrast to Playle's point source addition of this agent (Playle 1987). Previously, several workers had already added alum to small eutrophic lakes in the way we had done in an effort to improve water quality by removing phosphorus from the water bodies (Cooke and Kennedy 1978, Peterson et al. 1982, Welch et al. 1982).

In the autumn of the latter year, sediment removal and water exchange processes were applied in the Rowing pond. By removing the top sediment layer, the internal nutrient source can be reduced and the pond depth can be increased. This type of restoration measures has been taken in a number of smaller water bodies (Forsberg 1987).

In our investigations we applied chemical (pH, conductivity, cations, anions, orthophosphate, total phosphorus, NH_3, NO_2, NO_3, organic and total nitrogen, COD, etc.) analyses; biological (chlorophyll-a, phytoplankton, zooplankton) methods and statistical (similarity, diversity) evaluations.

In this paper we present the results from 1974, a year before the restoration, and the findings from 1984 and 1986 after the restoration, focussing mainly on the data of July.

RESULTS AND DISCUSSION

The conductivity values measured in 1984 and 1986 were ten times lower than in the period of pretreatment due to water exchange (Fig. 2). A drastic change could be observed in the halobity and ionic character of the original water quality (Table 2).

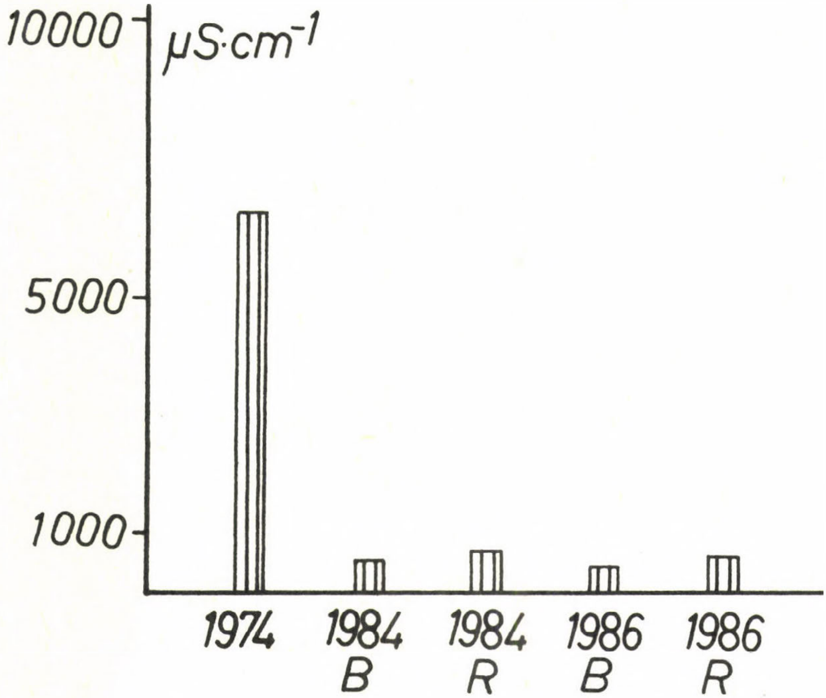


Fig. 2. Conductivity values measured in July

Table 2. Halobity characters of water bodies and their figured code (Felföldy 1976)

Year	Scale for halobity	Limno type	Ionic type
1974	Polyhalobic (9)	alpha-limno	Na-Cl
1984	Alpha-oligohalobic (3)	alpha-limno	$Na-CHO_3$
1986	Beta-alpha-oligohalobic (2)	alpha-limno	$Na-HCO_3$

The polyhalobic and Na-Cl character of the original water became alpha-oligohalobic by 1984 and two years later a beta-alpha-oligohalobic and a $Na\text{-}HCO_3$ character appeared in both ponds.

A very significant decrease in the orthophosphate (Fig. 3) and total phosphorus (Fig. 4) concentrations occurred after the exchange of water in both ponds. As a result of alum treatment in the Bathing pond the concentrations of phosphorus continued to decrease.

These results supported our goal to bind soluble orthophosphate with flocculation treatment applying alum, thereby decreasing its availability for using phytoplankton. In the Rowing pond this decreasing tendency could not be experienced.

Similar to the total phosphorus concentrations, the total nitrogen values were also decreasing considerably during the

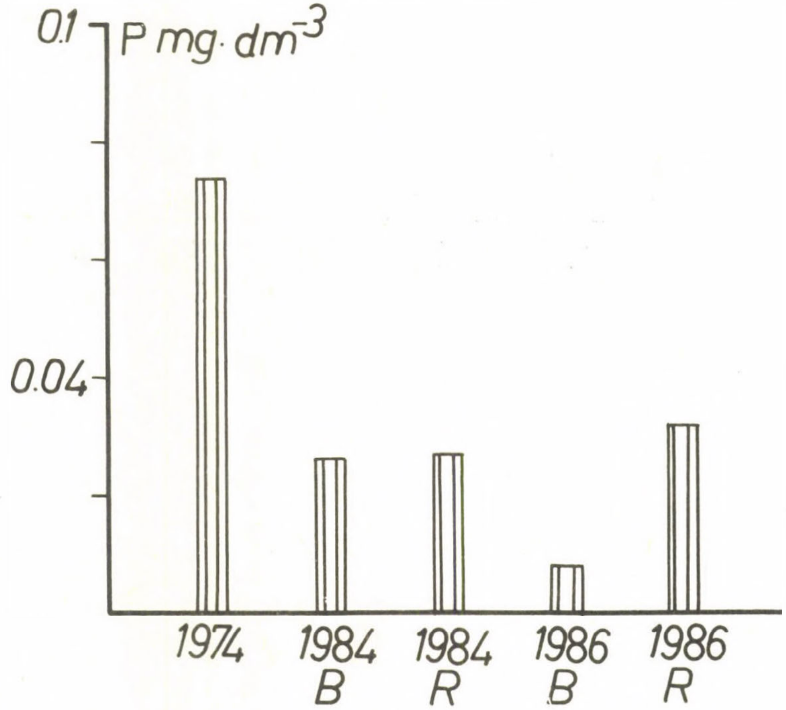


Fig. 3. Orthophosphate concentrations measured in July

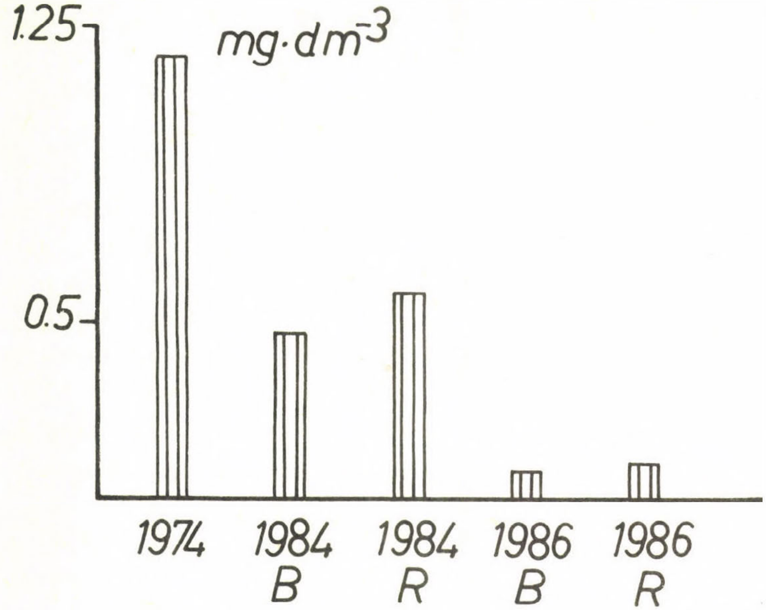


Fig. 4. Total phosphorus concentrations measured in July

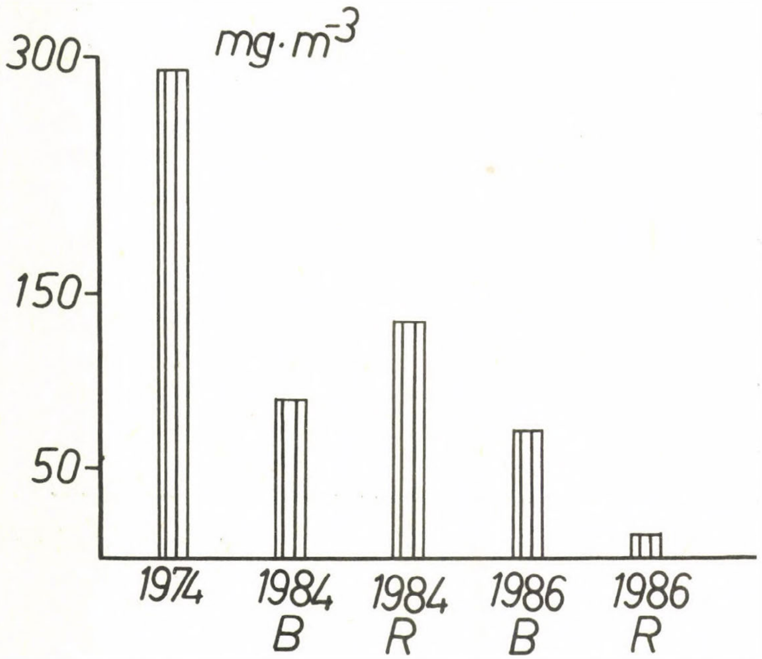


Fig. 5. Chlorophyll-a concentrations measured in July

study period, especially after the restoration processes in 1984.

A considerable reduction of chlorophyll-a content in the plankton could be observed (Fig. 5). This leads us to believe that the internal nutrient loading of the water bodies has altered, that is, decreased. For example, in the Bathing pond the decrease of the pigment concentration was fivefold.

In some lakes in which alum addition resulted in phosphorus reduction, substantial reductions in summer planktonic chlorophyll-a were observed (Cooke 1979, Foy and Fitzsimons 1987, Soltero et al. 1981).

The seasonal fluctuation of the ratio of total-N and chlorophyll-a content was different in the three years (Fig. 6).

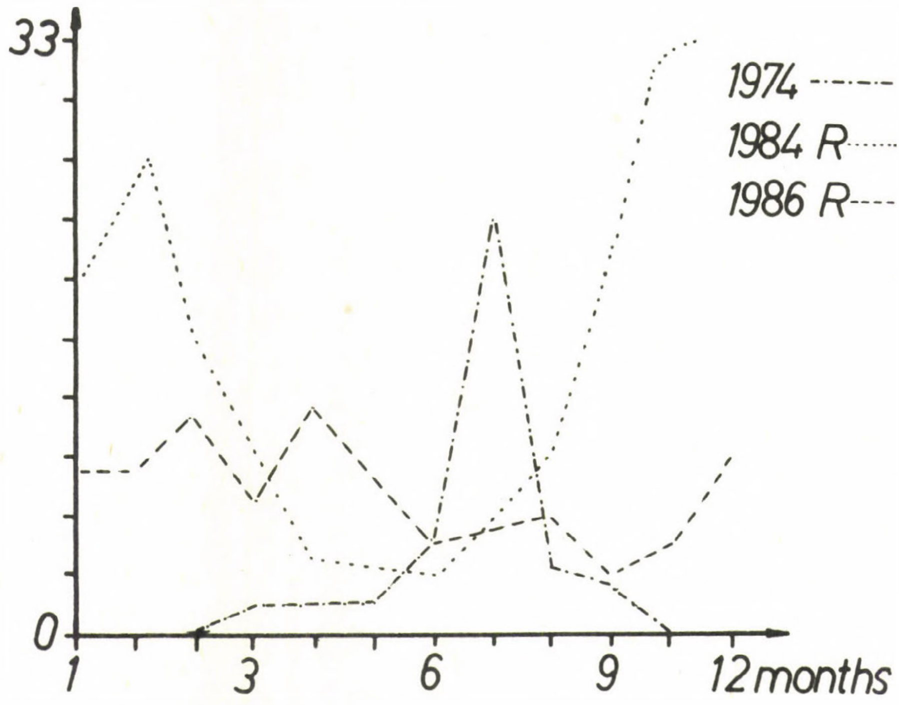


Fig. 6. Annual change in the ratio of total-N and chlorophyll-a content

In 1974 the maximal N supply had appeared in summer, but ten years later, as an interesting phenomenon, it shifted to the winter months which can be attributed to the decomposition of a reed grass species (milfoil, Myriophyllum spicatum). Goldman and Horne (1983) reported on the role of macrophytes in releasing phosphate from the sediment.

In 1986 no considerable peak could be measured and the values of the ratio were relatively low and almost the same all the year around. Considering the N and P ratio, and the total P and chlorophyll-a ratio, it can be stated that the water was abundant in phosphorus before the restoration treatment. After the restoration, however, phosphorus has become the limiting factor.

In Table 3, summarizing the results of phytoplankton analyses in the Bathing pond, beside the percentage distribution of different phylla, the total individual number of algae is also displayed.

Table 3. Results of phytoplankton analyses

Year	Phylla	Per cent	Total algal count 10^6 ind $\cdot$ l^{-1}
1974	Cyanophyta	40	
	Chlorophyta	60	468
1984	Cyanophyta	50	
	Euglenophyta	13	
	Chlorophyta	37	27
1986	Euglenophyta	25	
	Bacillariophyceae	25	
	Chlorophyta	50	6

Before restoration, phytoplankton mainly consisted of Cyanophyta and Chlorophyta species. In the samples collected in 1984 Euglenophyta species also occurred. Two years later, however, the Cyanophyta species disappeared. It is in good agreement with the statement of Foy and Fitzsimons (1987) that the direct addition of ferric aluminium sulphate resulted in a significant reduction of the individual number of the Cyanophyta species Oscillatoria agardhii var. isothrix. In phytoplankton several genera of Diatomae could be identified. First in 1984, the decrease of the individual number was twentyfold, but later it became seventyfold due to the change in the nitrogen and phosphorus supply (Fig. 6).

Similar to phytoplankton, the composition of zooplankton was also affected considerably. Originally, in zooplankton only Rotifera species were present; the total individual number was very high and the species diversity (Shannon's diversity) was low.

After the treatment, zooplankton became more diverse and poorer in individuals, corresponding to the findings of Knapp and Soltero (1983). In 1984 Cladocera species appeared. The dendrograms obtained by cluster analyses (Czekanowski's similarity index) based on the seasonal composition of phytoplankton (Fig. 7) and zooplankton (Fig. 8), respectively, demonstrate a similar arrangement of samples.

Samples taken in 1974 form a well-separated group, especially on the basis of the phytoplankton findings. In addition, the group of samples collected in 1986 is also differentiated from the other ones.

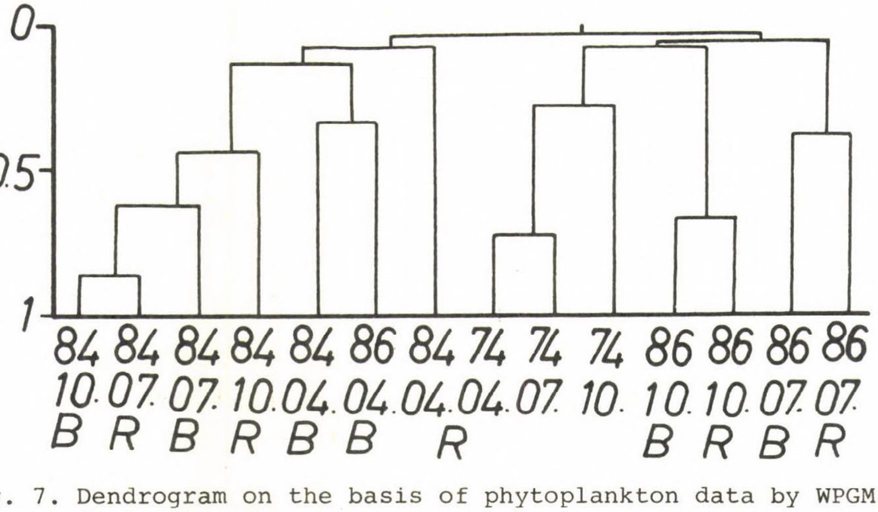


Fig. 7. Dendrogram on the basis of phytoplankton data by WPGM fusion

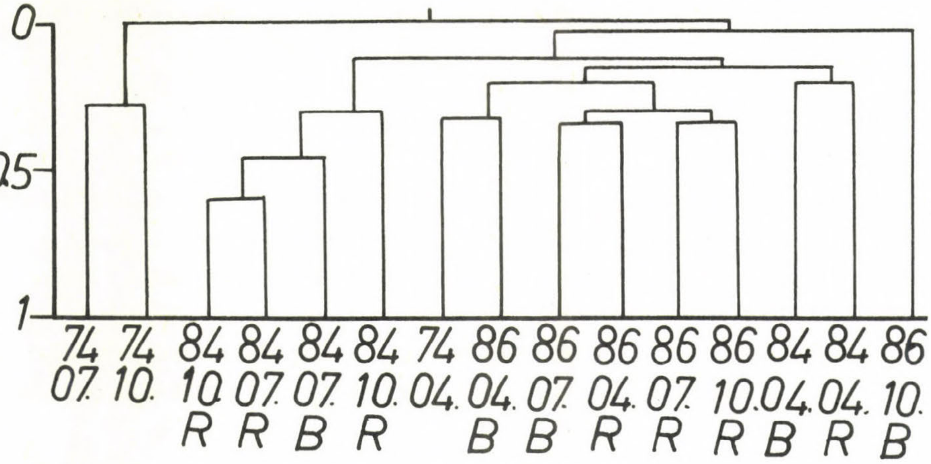


Fig. 8. Dendrogram on the basis of zooplankton data by WPGM fusion

Summing up our more important findings, we emphasize that the hypertrophic condition of the original SP became eutrophic in the Bathing pond and poly-eutrophic in the Rowing pond by 1984. Newer restoration processes in 1984 induced further favourable changes in water quality, and mesotrophic conditions developed in both ponds by 1986. Furthermore the favourable alterations of saprobity could also be established as a result of the restoration measures and the original polysaprobic conditions were changed into alpha-mesosaprobic.

In conclusion, the restoration processes applied in this case resulted in a better water quality during the study period. We think it expedient to maintain the ponds by regularly checking and measuring the water quality paramaters in order that the present and future of these ponds should develop satisfactorily.

REFERENCES

Björk, S. (1985): Scandinavian lake restoration activities. Proceedings Reprint, pp. 293-301.

Cooke, G.D. (1979): Evaluation of aluminium sulphate for phosphorus control in eutrophic lakes. OWRT Project No.

A-053. OHIO. Final Report. Ohio Water Resources Center, Columbus.

Cooke, G.D. and Kennedy, R.H. (1978): Effects of a hypolimnetic application of aluminium sulphate to a eutrophic lake. Verh. Int. Ver. Limnol. 20, 486-489.

Dévai, Gy., Dévai, I. and Lakatos, Gy. (1978): Vízminőségvizsgálatok északkelet-magyarországi víztereken (Water quality investigations in water bodies in Nort-East Hungary). Acta Biol. Debrecina 15, 51-89.

Dévai, Gy., Kollár, Gy. and Lakatos, Gy. (1979): Water quality deterioration and restoration in Pond "Sóstó" (NE-Hungary). Symp. Biol. Hung. 19, 113-120.

Felföldy, L.J.M. (1976): Biological water quality. A new system for the biological qualification of water. Research in water quality and water technology, Budapest 3, 1-37.

Forsberg, C. (1987): Evaluation of lake restoration in Sweden. Schweiz. Z. Hydrol. 49, 260-274.

Foy, R.H. and Fitzsimons, A.G. (1987): Phosphorus inactivation in a eutrophic lake by the direct addition of ferric aluminium sulphate: changes in phytoplankton populations. Freshwater Biology 17, 1-13.

Goldman, C.R. and Horne, A.J. (1983): Limnology. International Student Edition. McGraw-Hill, International Book Company, London, pp. 1-464.

Gächter, R. (1987): Lake restoration. Why oxygenation and artificial mixing cannot substitute for a decrease in the external phosphorus loading. Schweiz. Z. Hydrol. 49, 170-185.

Kollár, Gy., Öllős, G., Dévai, Gy. and Lakatos, Gy. (1978): A nyíregyházi Sóstó vízminőségromlásának okai és rehabilitációjának lehetőségei (The causes of water quality degradation in pond "Sóstó", NE-Hungary, and possibilities of its rehabilitation). Acta Biol. Debrecina 15, 101-141.

Knapp, S.M., Soltero, R.A. (1983): Trout-zooplankton relationships in Medical Lake, WA, following restoration by sulfate treatment. Journal of Freshwater Ecology 2, 1-12.

Lakatos, Gy. (1978): Comparative analysis of biotecton/periphyton samples collected from natural substrate in waters of different trophic state. Acta Bot. Acad. Sci. Hung. 24, 285-299.

Lerman, A. and Hull, A.B. (1987): Background aspects of lake restoration: water balance, heavy metal content, phosphorus homeostasis. Schweiz. Z. Hydrol. 49, 148-169.

Peterson, S.P. (1982): Lake restoration by sediment removal. Water Resources Bulletin 18, 423-436.

Playle, R.C. (1987): Chemical effects of spring and summer alum additions to a small, northwestern Ontario lake. Water, Air and Soil Pollution 34, 207-225.

Schindler, D.W. (1985): Coupling of elemental cycles by organisms: Evidence from whole-lake chemical perturbations. In: Stumm, W. (ed.): Chemical Processes in Lakes. Wiley, New York, pp. 225-250.

Vollenweider, R.A. (1968): The scientific basis of lake and eutrophication. Technical Report DAS/CSI/68, The Organization for Economic Cooperation and Development (OECD), Paris.

Welch, E.B., Michaud, J.P. and Perkins, M.A. (1982): Alum control of internal loading in a shallow lake. Water Resources Bulletin 18, 929-936.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest
Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

CHANGES IN TROPHY LEVEL INDICES OCCURRING DURING SEVERAL YEARS IN SOME SMALL LAKES OF POLAND

A. GIZIŃSKI

Department of Hydrobiology, Institute of Biology,
University of N. Copernicus
ul. Gagarina 9, 87-100 Toruń, Poland

INTRODUCTION

The basis of the preservation and management of lakes is the proper evaluation of their actual trophy level. This paper is based on the investigations of the author and his co-workers on the hydrochemistry, zooplankton and zoobenthos of the lakes in northern Poland. Some of them from among over 30 lakes under study were investigated for 4-8 years, with different regularity.

The chief aim of the paper was to recognize the range of several years' changes of some abiotic and biotic features (parameters, indices) of the lakes and to compare them in different lakes.

DESCRIPTION OF THE AREA AND METHODS OF STUDY

The lakes under study are located in the Tuchola Forest region (Bory Tucholskie, Bydgoszcz District). There are no superficial inflows and outflows. Some morphometric features of the lakes are as follows:

Lake	Area, ha	Depth, m max.	Depth, m mean	Type of mixis	I.m.*
Nawionek	11	9.8	6.3	dimictic	0.45
Czarne	9	7.8	4.3	polymictic	0.68
Gacno Wlk.	13	6.1	3.3	polymictic	0.87
Zmarłe	30	19.4	9.3	dimictic	0.40

*"Index of mixing" = $\frac{4.4\sqrt{D}}{\text{mean depth}}$ (Giziński 1978), where $4.4\sqrt{D}$ is the depth of epilimnion in metres (Patalas 1960).

The first three basins are the so-called lobelia lakes (Lobelia, Isoëtes), of low calcium concentration. In Lakes Czarne and Gacno the convectional polymixis was noted, caused not by wind action but by water heating from the bottom during calm, sunny weather (transparence usually down to the black, muddy bottom). All the lakes are located in the same forest region without any direct anthropopression.

Samples were taken at least 4 times per year. Standard methods were used. More detailed information on the hydrochemistry, zooplankton and zoobenthos of over 30 Polish lakes is presented by Giziński et al. (in press).

RESULTS AND DISCUSSION

The results of exemplary physical and hydrochemical observations are presented in Table 1.

Table 1. Several years' changes of some abiotic indices and parameters in the lakes under study

Lake	Year	TSI* (SD)	$P-PO_4$ mg dm^{-3}	N mineral. mg dm^{-3}	N mineral. / $P-PO_4$
Nawionek	1977	34.2	0.19	0.23	1.2
	78	36.2	0.24	0.25	1.1
	82	33.5	0.08	0.38	4.9
Czarne	1977	30.4	0.29	0.87	3.0
	78	34.6	0.35	0.62	1.8
	79	30.4	0.44	0.93	2.1
	80	-	0.06	0.25	4.5
	81	30.4	0.05	0.47	4.9
	82	33.9	0.06	0.08	1.4
Gacno	1977	31.9	0.30	0.23	0.8
	78	34.2	0.17	0.24	1.4
	79	31.9	0.45	0.34	0.8
Zmarłe	1977	39.6	0.33	0.78	2.4
	78	36.2	0.40	0.66	1.7
	79	36.8	0.43	0.65	1.5
	80	35.7	0.06	0.31	5.2
	81	34.6	0.08	0.73	9.0
	82	37.7	0.08	0.09	1.2
	83	-	0.07	0.07	1.0
	84	38.3	0.08	0.12	1.6

*TSI (SD) - after Carlson (1977)

The particular values shown in Table 1 were very different during the consecutive years. The direction of these changes was not the same in some lakes, e.g. in 1977-78 TSI (SD) values increased in three lakes and decreased in Lake Zmarłe. The N min./$P-PO_4$ ratio increased in Lake Gacno and dropped in the other three lakes.

The most distinct hydrochemical change was the rapid drop of $P-PO_4$ concentration in 1980. A similar phenomenon was noted in 1980 in other lakes, not described in this paper. It should be emphasized that precipitation in 1979 was low (400 mm $year^{-1}$) and in 1980 was twice as high (837 mm $year^{-1}$). Kentzer (1983) stated that concentration of biogenic salts in the groundwater inflowing to the lake was in 1980 higher than in the lake water. Therefore, the decrease of $P-PO_4$ content was not the result of lake water "dilution" during that rainy year. In our opinion, the most probable explanation of the decrease in phosphorus content is as follows:

Increase of precipitation → increase of near-bottom flow → destruction of near bottom oxygen microstratification → improvement of water oxygenation → sorption of phosphorus by the bottom deposits → decrease of soluble $P-PO_4$ in lake water.

It should also be mentioned that during the previous years processes of desorption prevailed in the water-mud interface (Kentzer, loc. cit.).

In Tables 2 and 3 some results of zooplankton and zoobenthos observations are presented. In both tables almost all values show an irregular variability. Some changes occurring in one lake often exceeded the differences recorded between individual lakes in one year.

Particularly interesting are the differences noted between Lakes Czarne and Gacno. These two lakes are very similar in their hydrochemistry, morphometry and in many other respects, considered not only in this paper. Some changes in these two lakes were of opposite direction during the same years (see values in Tables 2 and 3). Most variables were the values of the N max./N min. index illustrating the differences in seasonal changes of the zooplankton and bottom fauna.

Table 2. Several years' changes of some zooplankton parameters and indices

$\bar{N}$ - mean number, ind. dm^{-3}
NS - number of species (total zoopl.)
Rot. - Rotatoria
Crust. - Crustacea

Lake	Year	$\bar{N}$	$\frac{N\ max.}{N\ min.}$	$\frac{N\ Rot.}{N\ Crust.}$	NS	H'
Nawionek	1977	182	5	1.7	28	2.7
	82	141	3.5	3.5	22	-
Czarne	1977	47	6	0.7	24	2.0
	78	88	215	0.3	30	2.7
	82	200	-	9.0	22	-
Gacno	1977	78	158	124	15	0.7
	78	40	27	0.5	18	1.5
Zmarłe	1977	300	15	12.5	36	4.2
	79	172	5	1.8	40	3.7
	82	244	4	1.4	46	2.2
	83	197	2	3.9	36	3.3

Table 3. Several years' changes of some zoobenthal parameters and indices

$\bar{N}$ - mean number, ind. m^{-2}
NS - number of Chironomidae species
$\bar{B}$ - mean wet biomass, total zoobenthos, g m^{-2}
MW - mean weight of one Chironomidae larva, mg

Lake	Year	$\bar{N}$	$\frac{N\ max.}{N\ min.}$	$\bar{B}$	NS	MW
Nawionek	1977	700	15	1.2	19	1.5
	82	250	26	1.2	18	1.7
	84	450	4	0.9	13	1.3
Czarne	1977	2,500	4	5.4	10	1.2
	78	1,300	15	3.3	15	1.9
	79	300	8	1.3	6	6.9
	82	1,100	7	3.8	6	2.5
	84	450	3	1.7	12	3.4
Gacno	1977	1,500	13	2.9	10	0.9
	78	1,200	3	4.1	14	2.3
	79	3,000	3	7.8	10	1.9
	82	2,400	2	22.1	7	3.3
Zmarłe	1977	550	6	1.3	16	1.4
	78	1,100	4	4.9	17	1.5
	82	500	5	1.6	4	3.1
	84	400	2	2.0	7	8.9

Comparison of the data shown in Tables 1-3 allows to draw the following conclusions:

No correlations were found between the hydrochemical changes and those of the zooplankton and bottom fauna character. During the relatively stable hydrochemical regime of the lakes studied in 1977-79 the character of zooplankton and zoobenthos exhibited distinct changes (fluctuations). After 1979 drastic hydrochemical changes occurred in the lakes. At the same time, the faunistic changes, especially in the bottom fauna, were not as sharp as expected. Kentzer (1983) stated that in small lakes qualitative-quantitative changes in the bottom fauna occurred with 2-3 years' delay after the hydrochemical changes. This statement is not in agreement with the view that the bottom fauna is an indicator of the actual state of lake trophy.

CONCLUSIONS

1. The several-year hydrochemical changes of small lakes depend upon climatic-hydrological fluctuations.
2. Concerning the great variability of trophy level symptoms shown in this work, the trophic state of such lakes can only be determined on the basis of one-year investigations.
3. The course of several years' fluctuations is very different even in similar lakes.
4. The investigation results of the present author and others indicated a sharp "individualism" of the lakes concerning their long-term fluctuations, especially in the course of seasonal changes.

REFERENCES

Carlson, R.E. (1977): A trophic state index for lakes. Limnol. Oceanogr. 22, 2: 361-369.

Giziński, A. (1978): Significance of benthal fauna as indicator of eutrofication degree in lakes. Verh. Internat. Verein. Limnol. 20, 997-999.

Giziński, A., L. Błędzki, A. Kentzer, R. Wiśniewski, W. Zawiślak, J. Żbikowski, R. Żytkowicz (in press): Abiotic and zoocenotic conditions for the functioning of selected lake ecosystems with various trophic level. In: R. Bohr, M. Rejewski (eds): Some Ecological Processes of the Biological Systems in North Poland. Wyd. UAM, Poznań.

Kentzer, A. (1983): Wieloletnie zmiany fauny dennej wybranych jezior Tucholskich na tle zmian ich właściwości hydrochemicznych (Long-term changes of bottom fauna of selected Tuchola lakes based on their hydrochemical properties). Ph. D. Thesis, Dept. of Hydrobiology, Inst. of Biology UMK, Toruń.

Kentzer, A. (1983): The chemistry of ground waters inflowing to the selected lakes of Tuchola Forest Region (ibid.).

Patalas, K. (1960): Mieszanie wody jako czynnik okreálający intensywność krążenia materii w różnych morfologicznie jeziorach okolic Węgorzewa (Mixing of water as a factor determining intensity of matter circulation in morphologically different lakes of the Węgorzewo region) Roczn. Nauk Roln. 77-B, 1: 223-242.

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

SEDIMENT OXYGEN DEMAND IN WŁOCŁAWEK DAM RESERVOIR AND IN TWO DIFFERENT TROPHY LAKES: ROLE OF RESUSPENSION

R. WIŚNIEWSKI

Department of Zoology, Institute of Biology,
University of N. Copernicus
ul. Gagarina 9, 87-100 Toruń, Poland

INTRODUCTION

Resuspension of bottom deposits is an important factor in the cycling of fine-grain sediment, its redeposition and accumulation in water reservoirs. This process can locally modify the chemistry of water and influence its biology through release of nutrients and toxicants to the overlying water.

Under natural conditions, the main factors, which may initiate resuspension are wind stress and water movement. Therefore, particularly frequent resuspension can be observed either in shallow reservoirs or in the shallowy zones of deep reservoirs, as well as in water bodies of high water dynamics, like dam reservoirs and estuaries (Hunding 1979, Poltz 1980, Nichols 1986).

Resuspension exerts direct physical and chemical influence on hydrobionts through a considerable increase of the suspended matter concentration and, as a result of reduction of light and release of toxicants (Bruton 1985). It also causes an intensive increase of the sediment oxygen demand (Edwards and Rolley 1965, Boynton et al. 1981).

Recently, another resuspension factor has become important: human activity related to regulation and recultivation of water bodies. Dredging of water courses causes, in consequence, transposition of fine deposits. Regulation processes in the drainage area of a lake may lead to a drop-down of water level in the lake and thus expose fine sediment to wave activity in the accumulation zones. If the area of such activity is large enough it may affect oxygen balance in the whole lake through intensive sediment oxygen demand (Wiśniewski, in press).

It seems important to find unified measuring methods and to compare sediment oxygen demand (SOD) and sediment oxygen demand during resuspension (rSOD) in different water reservoirs. There are several laboratory and in situ methods for such measurements (Zeitzschel and Davies 1978, Bowman and Delfino 1980). Many of them differ in the kind of water used, i.e. lake water, tap water, oxygen content, temperature and time of incubation, water volume to sediment surface ratio, etc.

The aim of this work was to devise methods and tools for comparison of the results obtained in different water reservoirs.

MATERIAL AND METHODS

The investigations were conducted in three reservoirs differing in morphology and trophy (Table 1 and Fig. 1).

Table 1. Some characteristics of the reservoirs

	Włocławek Dam Reservoir	Lake Partęczyny	Lake Druzno
Area (km^2)	75.0	3.2	12.6-20.9
Length (km)	57.0	4.3	10.0
Breadth (km)	2.5	1.1	2.2
Depth max. (m)	15.5	28.5	3.0
mean (m)	5.5	6.8	1.2

In Włocławek Dam Reservoir special attention was paid to its central part. Earlier investigations showed that there are two zones with different hydrology in this part of the reservoir. The first one is deep (11 m), relatively narrow, with a fast water current, and the second one is shallow (4-5 m), wide, exposed to wind which causes heavy waving.

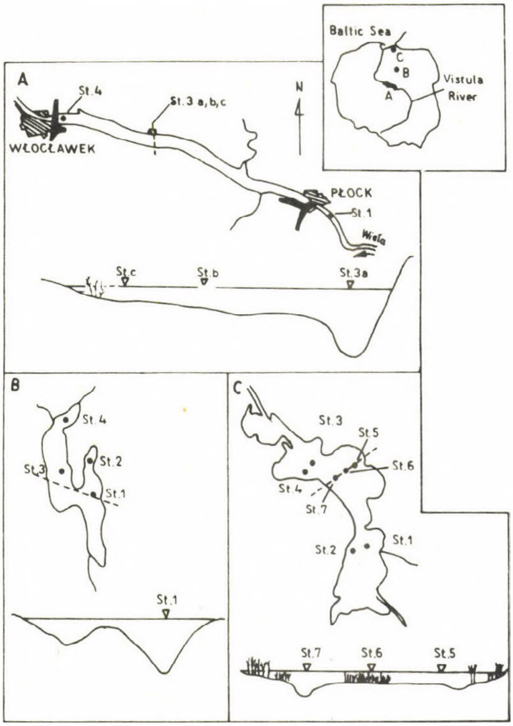


Fig. 1. Location of sampling sites. A - Włocławek Dam Reservoir, B - Lake Partęczyny, C - Lake Druzno.

In the moderately eutrophied Lake Partęczyny sampling stations were situated in the two deepest zones and in the two shallowest parts of the lake.

Lake Druzno is shallow, eutrophied, with a water-course and wide zones covered with macrophytes.

Samples of bottom deposits for laboratory measurements were taken with the Ekman and Kajak-type samplers. A 10 cm^3 volume of the upper layer sediment was placed in a 750 cm^3 bottle filled with distilled water aerated to approx. 100% saturation. In one series of experiments, deposits remained motionless; in the second series sediment was maintained in suspension by stirring with a magnetic stirrer or reversing the bottles. After incubation for 2-4 h, at 19-21^{o}C, 50 cm^3 water samples were taken for analysis of the oxygen content by the Winkler method.

During the in situ investigations, on shallow stations, PVC tubes were inserted into the deposits. In one series of measurements with undisturbed sediment SOD was estimated. In the second series with stirred sediment rSOD measurements were made.

On the deeper stations of up to 6 m depth, an originally constructed device was employed for respirometric measurements. The device can also serve as diffusive chamber for measurements of nutrient exchange in the mud-water interface (Fig. 2).

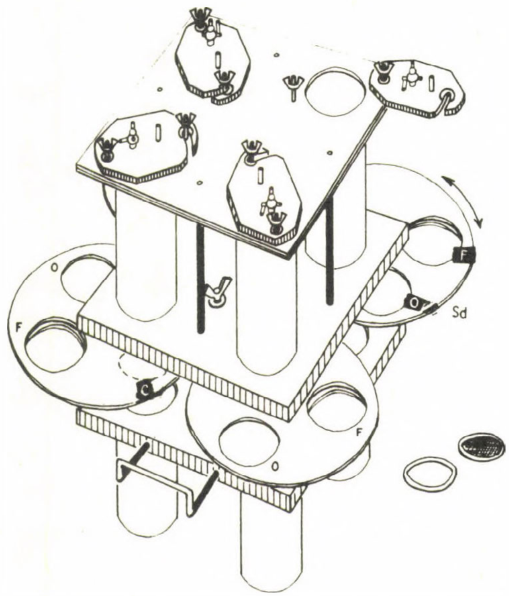


Fig. 2. Device for in situ SOD measurements. Sd - separating disc set in three possible positions: C - closed, F - filter holding, O - opened.

It consists of two parts separated with metaplex discs which can be set by turning in three desired positions - open, closed and filter holding. The lower part was inserted in the deposits by a diver and remained there before and during the measurements. The upper tubes were filled with natural water filtered through a Whatman GF/C filter as well as with distilled water containing the required oxygen concentration. The upper part was then attached to the lower part. Separating discs were set in the desired position and both parts were firmly screwed together. Samples of water for oxygen analysis were taken with a syringe.

The detailed description of the apparatus and its deep-water version will be published elsewhere (Wiśniewski, in prep.).

Resedimentation time analyses were made in 1 dm^3 measuring cylinders. Sample of 100 cm^3 of suspension was filtered through a GF/C filter, then dried and weighed.

RESULTS AND DISCUSSION

As shown in Table 2, SOD and rSOD differ depending on stations and sampling periods. Shallow stations had a distinctly lower SOD. Higher rSOD was noted at the deepest stations and in those situated in the zones of sediment accumulation: St. 3b in Włocławek Reservoir and St. 1 in Lake Druzno.

Figure 3 shows that in all the investigated reservoirs the deepest stations contained a very fine fraction of sediment capable of remaining in suspension for a relatively long time.

Results of laboratory and in situ experiments (Table 3) did not differ remarkably, though in situ measurements showed somewhat higher rSOD values.

Addition of an inhibitor (Table 4) caused a slight lowering of both SOD and rSOD in comparison to uninhibited sediments.

The obtained results allow an evaluation of the method used. Simultaneous measurements of SOD and rSOD can give information about the extent of SOD increase during resuspension in the zones of fine-grain sediment accumulation. Its ability to remain in suspension for a relatively long time can locally decrease oxygen content in the overlying water even if the stress factor is of short duration.

It seems that in comparing the results obtained in the different reservoirs, laboratory measurements have some advantages over in situ investigations. Factors affecting SOD (temperature, initial oxygen content, water movement intensity) can be easily controlled. However, placing a definite amount of sedi-

Table 2. Sediment oxygen demand (SOD) - upper value and sediment oxygen demand during resuspension (rSOD) - lower value in Włocławek Dam Reservoir, Lake Partęczyny and Lake Druzno (g m^{-2} day^{-1})

Station	Depth (m)	Organic matter (%)					
Włocławek Reservoir			21.05 1987	3.09	25.04 1988	9.05	4.08
1	3.0	0.7	9.3	11.5			
			9.8	13.8			
3a	11.0	12.0	1.8	2.7	8.4	2.9	4.7
			26.5	25.2	82.8	63.7	41.6
3b	5.5	12.2	3.1	4.5	8.3	1.9	4.7
			26.2	30.5	82.1	65.6	48.9
3c	2.0	15.8	2.9	5.0			
			23.5	30.5			
4	12.0	14.4	3.8	3.7			3.9
			29.7	30.5			52.7
Lake Partęczyny			27.08 1987	27.04 1988	17.05	26.08	
1	27.0	21.1	3.9	5.3	3.6	3.7	
			43.8	53.6	55.4	62.9	
2	5.0	27.1	3.9	0.9	0.2	1.9	
			13.2	13.0	22.2	14.7	
3	13.0	26.2	3.6	6.8	7.4	3.7	
			30.7	39.7	42.5	48.0	
4	5.5	34.2	2.2	6.5	1.8	1.9	
			22.6	27.7	31.4	22.2	
Lake Druzno			4.08 1987	7.09	30.04 1988	21.05	25.08
1	2.5	16.9	3.6	4.8	14.4	5.5	5.5
			40.8	45.6	55.4	45.6	48.0
2	0.5	14.9	1.2	2.4	15.2	5.5	1.9
			19.2	22.8	51.6	46.2	16.7
3	2.0	9.8	3.0	4.2	13.2	3.6	1.9
			36.0	38.4	48.0	44.3	38.7
4	0.5	10.8	2.4	4.2	10.8	0.2	9.2
			36.4	28.2	56.4	46.2	44.4

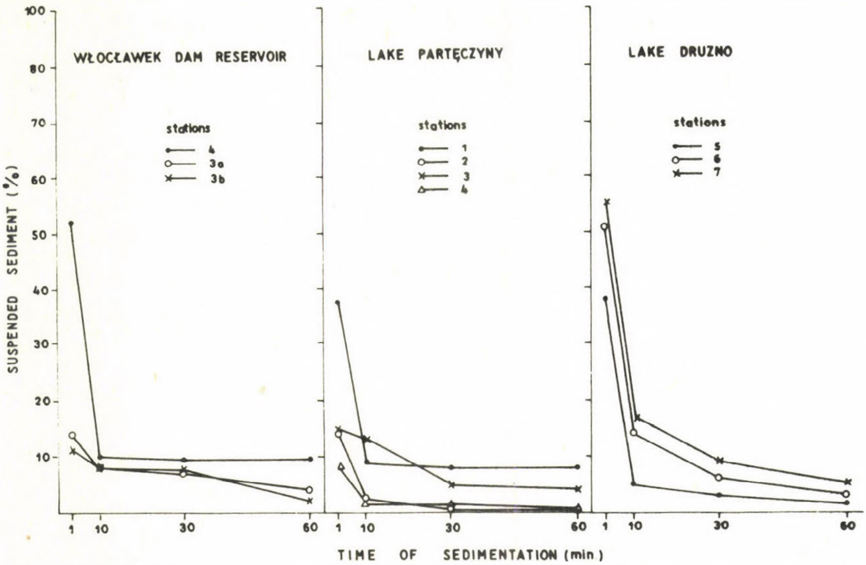


Fig. 3. Comparison of the sedimentation character of deposits from the investigated reservoirs.

Table 3. SOD - upper value and rSOD - lower value (g m^{-2} day^{-1}). Comparison of laboratory and in situ measurements. Incubation during 2 h at 19°C. Water filtered through Whatman GF/C filter

Sample	Laboratory	In situ	
	Distilled water	PVC tubes Unfiltered water	Respirometer Filtered water
L. Druzno St. 1	4.2 56.8	2.2 62.4	2.6 61.8
Włocławek St. 3b	1.6 48.2	1.0 52.4	1.4 54.4

ment, either by weight or by volume, into the incubation vessel usually disturbs the physical and chemical properties of the sediment. It is especially important in SOD measurements (Edberg and Hofsten 1973).

Table 4. Chemical oxygen demand (COD) share in SOD - upper value and in rSOD - lower value (g m^{-2} day^{-1}). Inhibitor - phenol (3 g 300 cm^3). Incubation during 2 h at 19°C

Sample	Distilled water			Lake water		
	uninhibited	inhibited	COD (%)	uninhibited	inhibited	COD (%)
L. Druzno	3.6	3.2	88.8	3.9	3.4	87.1
St. 1	45.6	43.8	96.0	48.0	44.2	92.1
L. Partęczyny	3.7	3.0	81.1	3.7	3.7	100.0
St. 1	55.4	53.6	96.7	62.9	59.2	94.1
Włocławek	4.2	3.8	89.4	3.9	3.2	82.0
Reservoir	51.3	48.4	94.3	52.7	47.9	90.9
St. 4						

The optimal way to avoid this problem could be the in situ measurements with standard water, e.g. reservoir water filtered and saturated to 100% oxygen solubility or distilled water. In case of distilled water the task is even easier due to the greater homogeneity of water, namely its physical and chemical properties, i.e. oxygen solubility, nutrient and ionic contents, viscosity, etc. However, there appears the disadvantage of hypotonicity to hydrobionts and the resultant error in SOD biological fraction count.

As shown in Table 4, the main constituent of SOD, especially of rSOD, in the investigated reservoirs was the chemical oxygen demand (90.1-96.7%). Wuncheng (1980) reported that in a shallow oxbow lake, COD can reach 71.0 to 100% of SOD, mainly due to ferrous demand.

It seems that for COD-dominated deposits, incubated for only a short time, which limits the biological fraction of SOD, the in situ respirometric method with oxygen-saturated distilled water can be useful.

The increase of SOD in time of resuspension expressed as a result of subtraction or SOD/rSOD ratio could be an index of potential sediment oxygen demand.

REFERENCES

Bowman, G.T. and Delfino, J.J. (1980): Sediment oxygen demand techniques: A review and comparison of laboratory and in situ systems. Water Research, 14, 491-499.

Boynton, W.R., Kemp, M.W., Osborne, C.G., Kaumeyer, K.R., Jenkins, M.C. (1981): Influence of water circulation rate on in situ measurements of benthic community respiration. Mar. Biol. 62, 185-190.

Bruton, M.N. (1985): The effects of suspensoids on fish. Hydrobiol. 125, 221-224.

Edberg, N. and Hofsten, B.V. (1973): Oxygen uptake of bottom sediment studies in situ and in the laboratory. Water Research. 7, 1285-1294.

Edwards, R.W. and Rolley, H.L.J. (1965): Oxygen consumption of river muds. J. Ecology 53, 1-19.

Hunding, C. (1979): The oxygen balance of Lake Myvatn, Iceland. Oikos 32, 139-150.

Poltz, J. (1980): Some studies on the problems of "Treibmudde" in Steinhuder Meer. Develop. Hydrobiol. 3, 3-9.

Wiśniewski, R.: Eutrophication of Lake Gopło. Faunistic response to hydrological changes. AUNC. Limnol. Pap. 17 (in print).

Wiśniewski, R.: A universal device for in situ sediment oxygen demand measurements (in prep.).

Wuncheng, W. (1980): Fractionation of sediment oxygen demand. Water Research 14, 603-612.

Zeitzschel, B. and Davies, J.M. (1978): Benthic growth chambers. Rapp. P.-v. Reun. Cons. int. Explor. Mer. 173, 31-42.

ACIDIFICATION AND TOXIC POLLUTANTS

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

GLOBAL STATE OF LAKE ACIDIFICATION AND ITS CONTROL

C. FORSBERG

Institute of Limnology, Uppsala University
Box 557, 751 22 Uppsala, Sweden

Abstract

This paper presents a brief survey of lake acidification of today including critical loads for sulfate deposition and control measures. Increasing importance of atmospheric nitrogen emissions and evidence of reversibility of lake acidification, and time aspects on the future development are discussed.

Introduction

In the 1960's, lake acidification was identified in areas of Scandinavia with thin soils and poor buffering capacity. The appearance of these acidified lakes was certainly a gradual and delayed response to decades of increasing deposition of acidifying air pollution. Thus, an estimate of sulphur emissions in Europe shows a gradual increase during the 20th century, with a very steep increase after 1950 (OECD 1977). After twenty years of rapidly increasing emissions of sulphur oxides, the amounts of these pollutants generally stabilized around 1970. In the eastern US the increase continued into the early 1980's, while in most parts of western Europe the emissions declined (ECE 1987a).
Acidification was initially viewed as a Norwegian and Swedish problem, but later nearly every European country was found to be affected (Wright 1983). Historical trends in lake water quality are difficult to describe as reliable water quality data are available only for a few decades at best. Reconstructions have had to be made by sediment analyses, which give examples of drastic decreases in lake pH from the1950's and 1960's (see e. g. Renberg & Hedberg 1982, ECE 1987a, Battarbee & Renberg 1987). Very limited knowledge exists regarding the time aspects of processes which steer the development of acidification, especially of those at the level of soil chemistry. This will of course hamper the discussion of the recovery processes, which are expected to appear when the acidifying emissions are being reduced.

The recent literature on lake acidification is vast, reflected e. g. by Schindler's (1988) comprehensive list of references and notes. In his article the central interest is paid to increasing the understanding of acid rain and its effect on North American waters, but it also includes a lot of central, and more general information.The aim of this paper is to give a brief global survey of lake acidification of today, evidence of its reversibility and efforts to control the acidifying emissions .

Regional estimates of acidified lakes

Acid rain has been known in Scandinavia and Northern America for about 20 years . Increased knowledge demonstrates that now acid rain is also widespread in many other parts of the world, and the extent of acid-sensitive areas is much larger than was believed a decade ago (Table 1).

Table 1. Distribution of acid rain and acid-sensitive areas. From Schindler (1988)

Acid rain observed in	Acid-sensitive areas
Europe	Scandinavia, UK, The Netherlands, Belgium, Denmark, Ireland, Italy, Switzerland, West Germany
Canada	All western provinces The Yukon, The Northwest Territories, Labrador
USA	Northeastern states, Minnesota, Wisconsin, upper Michigan, several southeastern states, western mountain areas
The Soviet Union	
South America	South America Africa
Asia	China, Japan

Lake acidification was initially viewed as a Scandinavian, Canadian and North American problem. Today many other regions in the US (see below) and Europe have acidified lakes or lakes reported as affected (Wright 1983, ECE 1987a, Landers et al. 1988). In The Netherlands, for instance, about 90% of the small lakes, moorland pools, ponds and dune pools have recently been acidified, about one third of them with pH <4.0 (Schuurkes 1987). Similar developments are reported for small lakes on sandy soils in Belgium (Vangenechten 1983). Acidified waters are reported also from East Germany, Denmark, Finland, West Germany and United Kingdom (Wright 1983).

Estimating the present extent of lakes affected by acid deposition will include a considerable uncertainty, as regional or national scale figures are extrapolations from site-specific studies (Landers et al.1988). However, in order to get some ideas of the numbers of threatened lakes, the estimated situation in Sweden and parts of the USA is presented in Table 2.

Table 2. Regional estimates of acidic and susceptible lakes

Region	Acidic lakes, %	Susceptible lakes, %	Reference
Sweden	5	41	Monitor 1986
Canada, 6 east. provinces	-	50	Schindler 1988
USA, The Northeast	6	60	Landers et al. 1988
" , The Upper Midwest	2	41	"
" , Florida	22	-	"
" , Adirondacks	11	36	Brakke et al. 1988
" , Poconos/Catskills		<50	"
" , Southern N.England	5	57	"
" , Upper Michigan	10	-	Eilers et al. 1988
" , Northcentral Wiscon.	5	41	"

In Sweden about 35,000 lakes, or 41% of the total number, are estimated to have winter pH values <6. They are here looked upon as susceptible to further acid deposition (Table 2). In about 21,500 lakes acid sensitive organisms such as roach and crayfish cannot survive. Loss of fish populations is expected in about 4,500 lakes, where pH values are about or below 4.9 (acidic lakes, Table 2). Most of these lakes are small. Anthropogenic air pollution seems to be responsible for the acidification of about 15,000 Swedish lakes, while almost 7,000 lakes were already acidic during pre-industrial time (Monitor 1986).

The US lakes listed as acidic are lakes which have an acid neutralizing capacity (ANC) ≤0, while the susceptible ones have an ANC ≤200 μequiv/l (Landers et al. 1988) or ANC ≤ 50 μequiv/l (Brakke et al. 1988, Eilers et al.

1988). The US material probably gives underestimated values as small lakes, generally those <4 ha up to about 10 ha, are not included (Landers et al. 1988). For example, the Adirondacks contain many small waters which if included should have risen the percentage of acidic lakes (Brakke et al.1988). As an approximation, 5-10% of Swedish and eastern North American lakes can be regarded as acidic, while 40-50% of the lakes are susceptible to further deposition of acidifying compounds.

Knowing the difficulties involved in modern society reducing acidifying emissions to permissible levels, an increasing number of acidic lakes can be expected in the near future, not only in Sweden and North America, but also in many other parts of the world.

Acidification of lakes on a regional basis corresponding closely with the pattern of atmospheric deposition of sulphur has been demonstrated for Scandinavian lakes (Wright & Henriksen 1978, Monitor 1986). Recently, the same principle pattern has also been demonstrated for Ontario, Canada (Neary & Dillon 1988). As sulphur deposition increased, in-lake concentration of sulfate increased, while alkalinity and pH- values decreased. The first regional probability sampling of lakes in the US also indicated that lake sulfate concentrations correlated highly with estimated sulfate deposition (Sullivan et al. 1988). Further support for clear links between atmospheric acid deposition and lake acidification is presented by Eilers et al. (1988), who related in-lake ANC to a gradient of acid deposition. Schindler (1988) in summarizing four lines of evidence, stated that acidification of lakes has occurred in geologically sensitive areas receiving acid polluted precipitation, but not in sensitive areas still having precipitation with relatively low acidity.

Critical loads for sulfate

The sensitivity of an ecosystem to acid deposition depends on many different factors which makes the assessment of critical loads very complicated. Nevertheless quantitative critical loads have been proposed based on empirical relationships between measured deposition and pH conditions, or ratios between alkalinity and Ca^{2+}+Mg^{2+} in lakes, or on process-oriented models (e. g., Henriksen et al. 1986, Kämäri 1986, Schindler 1988). Critical load has been defined here as " the highest load that will not cause chemical changes leading to long-term harmful effects on the most sensitive ecological systems" (Nilsson 1986). For surface waters the long-term was given to "within 50 years", which, however, seems to be rather short in this context. Critical load also means deposition values that will maintain pH levels above 5.3 in the most sensitive lakes. Furthermore, alkalinity production of the total catchment area should balance the effective acid loading, ensuring no dominance of strong acids (pH >5.3) in surface waters (op. cit.). The critical loads of sulfate are estimates based on certain assumptions, and do not apply to humic lakes. In some areas with large variations in the amount of precipitation, as in western Norway, concentrations in precipitation are recommended for developing estimates of critical loads (Henriksen & Brakke 1988).

Estimates of critical sulfate loads have been presented for North America, Norway and Sweden (Table 3). The higher figures for the lakes in the eas-

Table 3. Estimated critical loads of sulfate, kg/ha*yr (wet deposition)

Geographic area	Critical loads	Reference
Canada	~ 20	Henriksen et al. 1986
W. Norway	~ 20	"
S. and E. Norway	~10	"
Eastern USA	10 - 30	Henriksen & Brakke 1988
Sweden	~ 10	Nilsson 1986

tern USA is explained by a better buffering than the acidified Norwegian and Swedish lakes (op. cit.). In sensitive aquatic ecosystems, anthropogenic contributions of atmospheric sulphur must be very small to maintain positive alkalinity. An acceptable critical load on lakes in southern Norway has been estimated to correspond to 20% of the present deposition. Even at a 100% reduction there could still be some high altitude lakes without alkalinity. In a sample of about 1,000 Norwegian lakes, 3-7% were estimated to belong to this category. To prevent episodes critical to fish, even lower loads than the listed ones may be necessary (Henriksen et al. 1986).
To protect the most sensitive waters, the limit of wet SO_4^{2-} deposition must be somewhere between 9 to 14 kg/ha*yr, which is far below the present values of 20-50 kg/ha*yr currently observed in most of western Europe and North America (Schindler 1988). The difference between the present loads and the estimated critical ones demonstrates that the reduction of anthropogenic sulfate deposition must be very effective to prevent further lake acidification. From the figures given by Dickson (1986), the reduction must be 75-85%.

Increasing importance of atmospheric nitrogen emissions

Emissions of nitrogen oxides are estimated to undergo major increases over the next two decades in both Europe and North America (ECE 1984). The effects of this increase were demonstrated by the observed increase in precipitation acidity by NO3, which counteracted expected positive effects of decreases in SO2 emissions (Hedin et al. 1987). Therefore, increases in lake acidification can be expected, in spite of some reductions in SO_2 emissions (ECE 1987a). Inputs of nitric acid may already have contributed to long term acidification in the Adirondack region (Driscoll & Schafran 1984). In many areas atmospheric ammonia deposition is enhanced, and its importance in acidification processes is gaining increasing attention (see e.g., Shuurkes 1987). Thus, the role of nitrogen as an acidifying agent is expected to increase, both absolutely and relatively (Hauhs 1987).

In most of Europe most of the deposited nitrogen compounds are still stored in the soil (Dickson 1986). When setting critical loads for nitrogen, the aim has been mainly to define deposition levels which will prevent forest ecosystems to become nitrogen-saturated and thereby prevent nitrogen from leaching to surface waters. For many coniferous ecosystems, the critical load is in the range of 10-20 kg N/ha*yr, in terms of total deposition (Nilsson 1986). Exceeding this level means leakage of nitrogen, which has already begun in large areas. Thus, increasing leaching of nitrate is reported from Scandinavia where the total deposition exceeds 15 kg N/ha*yr (Dickson 1986).

Even at loading levels much below the range mentioned above, an increasing leaching of nitrogen, mainly as organic-N, has recently been demonstrated for the Swedish forest river Dalälven (Forsberg & Löfgren 1988). During the last twenty years, concentrations of total-N and organic/humic material have increased significantly (Tot-N by 6 µg/l*yr, of which 5 was organic-N). River transports of nitrogen and humic material (including org-N) showed a pattern of accelerated increase during the last 15 years, while that of PO4-P decreased at the same time as transport of Tot-P was unchanged. These results may indicate that the increased atmospheric-N fallout together with PO4-P from the catchment have increased the basically N-limited forest primary production, resulting in litter increase. Microbial decomposition of this increased amount of litter is a likely explanation to the increasing transport of humic material (Forsberg 1988).

Leakage of humic material seems to be a large scale phenomenon in Sweden, with average increase in transports of 2 to 3% per year for forest rivers in central Sweden (Ahl 1988). The increased leakage of humic material may influence many chemical and biological processes in aquatic ecosystems, e. g., acidification, algal growth and transports of heavy metals.

Control measures

Control of lake acidification as well as of other damages caused by acid deposition is mainly a question of reducing acidifying atmospheric emissions. Results from the major review in 1986 prepared within the framework of the Convention on Long-range Transboundary Air Pollution, demonstrate the present national strategies and policies for the abatement of air pollution (ECE 1987). Most countries have reduced their sulphur dioxide emissions compared to 1980, 10 of them by 30% or more. In 1995 all 31 parties estimated their emissions to be 30% or more below the 1980 level, 11 of them forecasted a reduction by about 50% or more. In Europe, the UK is still one of the few countries refusing to sign the "30 percent club" Protocol calling for a 30% reduction in sulphur between 1980 and 1993 (Barnaby 1988).

Emissions of nitrogen oxides were mostly unchanged over the past three to five years. Estimated trends to 1995 varied from an increase by 50% to a decrease by the same figure. Eight countries planned to reduce their nitrogen emission by 1995. It was concluded that further emission reductions of sulphur are required and greater attention for action to reduce nitrogen oxides is needed (ECE 1987b).

Lake liming was introduced in Sweden about 10 years ago as a temporary defense strategy to control acid lakes until emission reductions have been carried out. Grants from the Government support local, regional and private liming programs with a grant for 1987 of SEK 100 million. Available Norwegian liming subsidies that year were NOK 11 million. Both positive and negative effects of lake liming are reported. Liming operations on land are recommended in order to reduce the negative effects from the acidified catchment area (Dicksson 1987). New and comprehensive studies on the effects of lake liming have recently been presented (Dickson (ed.) 1988) from the earlier well documented acidified lake Gårdsjön, situated on the Swedish west coast (Andersson & Olsson,1985). Dickson (op. cit) concluded that lake liming alone could not save lakes from damages caused by acids and metals from the catchment area.

Reversibility of lake acidification

The question of reversibility of lake acidification is of the most central interest, e. g., when discussing the measures necessary to take to reduce the acidifying emissions to the atmosphere. However, today there is very limited knowledge about the concept of lake recovery, illustratively demonstrated by the first book on this topic (Barth 1987, ed.). Only six of a total of 16 papers deal with the recovery concept, which means that the title of this book " Reversibility of Acidification", is somewhat misleading.

The acidification and deacidification of lakes depend to a high degree on different soil processes. There is a lack of experience about the time perspectives of both these processes. In contrast to the development of acidification which can be reconstructed from sediment studies (Battarbee & Renberg 1988), it is still too early, or impossible, to get more general information about recovery by reconstructions or by empirical evidence. Therefore, experimental studies or modelling have to be used when trying to predict recovery of acidic lakes after reductions of acidifying emission. Examples of two models are given in Table 4 together with empirical and experimental evidence.

The most pronounced reductions so far are reported from the Sudbury area (Canada) where SO_2 emissions were reduced by about 50%. This reduction resulted in recovery in lake SO_4 (decreases by 25 to 44%), pH (increases from 0.4 to 1.7 units) and Al (Dillon et al. 1986). Empirical evidence demonstrates for the Canadian lakes in the Sudbury area that lake recoveries occurred more or less at rates predicted by changes in deposition and water residence time (Schindler 1987, 1988). Schindler (1987) concluded optimistically that after a sufficient reduction in deposition of acidifying compounds, lakes will rapidly become chemically suitable for their natural biota, but also that it is uncertain whether lakes will recover completely (Schindler 1988). However, evidence from this area having large proportions of bare rocks will have only limited relevance for systems where water to a higher degree percolates through soils before reaching the lakes (Hauhs 1987).

Another example of partial recovery is reported from the Swedish westcoast area, where wet deposition of SO_4 had decreased by about 20%. Reduction in lake sulfate and small improvements of the pH-values in some lakes were observed (0.3 to 0.4 units at most). The response to changes during "deacidification" was hysteretic (Forsberg et al. 1985) and sulfate lost part of its earlier more central role in steering the ionic composition of the lake water (Forsberg & Morling 1988).

The experimental and spectacular RAIN project (Reversing Acidification In Norway) which started in 1984, is now generating interesting results. Substituting acid rain by clean rain lowered the concentrations of the strong-acid anions NO_3 from 35 to 7 μequiv/l and SO_4 from 110 to 53 μequiv/l in the

Table 4. Evidence of reversibility of lake acidification

Evidence	Results	Reference
Empirical	Sudbury lakes: Partial recovery	Dillon et al. 1986, Schindler 1987
"	Swedish westcoast lakes: Partial recovery	Forsberg et al. 1985.
Experi-mental	RAIN project, Norway: Indications of recovery	Wright et al. 1988
"	Greenhouse experiments: Indications of recovery	Schuurkes 1987
By models	MAGIC model: Simulation of reversibility	Cosby 1987
"	RAINS model: Simulation of reversibility	Alcamo et al. 1987

runoff compared to those of the controls. By increased dissociation the organic acids became increasingly important. Only small increases in pH of runoff occurred, explained by buffering of organic acids. The experiment shows that chemical changes caused by acid rain are largely reversible. After a major reduction in the flux of strong-acid anions, major positive changes in pH and inorganic aluminium in clearwaters in southern Norway can be expected (Wright et al. 1988, see also Wright 1987).
Greenhouse experiments with small-scale aquatic ecosystems showed that reversibility of lake acidification may be possible, but only after severe reduction of atmospheric deposition of acid . The question of re-establishment of the former fauna and flora is stressed (Schuurkes 1987).

A number of process-oriented mathematical models has been developed to simulate acidic input, soil system processes and resulting chemical surface water response. One of the most widely used is the MAGIC model (Model of

Acidification of Groundwater In Catchments), a process-oriented model which can be used to simulate chemical reversibility (Cosby 1987). This model, when applied to the first two years of data from the RAIN project, generally predicted the actual changes observed (Wright et al. 1988). The other model mentioned in Table 4, namely the RAINS (Regional Acidification INformation and Simulation) model is designed as a tool for evaluating control strategies. It is now sulphur-based, but is being modified to also include nitrogen species (Alcamo et al. 1987).
Due to the complexity of soil-water systems, these models have, of course, inherent uncertainties which limit their value as tools for evaluating the response of acidic lakes to emission reduction measures. In addition, predictions of long-term responses are very difficult to verify, which means that these predictions are still of limited value (Kämäri 1986, Cosby 1987).

Discussion - time aspects

Time aspects on the future development of lake acidification can be separated into two parts. The first includes man and deals with the primary question of when the necessary reductions of acidifying emissions can be carried out? The second part is then to what degree can soil systems/lakes recover and how long will it take?
If and when modern industrialized society will attain the permissible loading levels which will allow lakes to recover is still an open question. Most countries in the "30% club" have reduced their sulphur dioxide emissions compared to 1980. But still all countries are far from the reductions of 80-100%, which are required for lakes to recover in the acid sensitive areas . Emissions of nitrogen oxides are predicted to both increase and decrease, and no international agreements on reductions of NOx emissions have been achieved. The emission situation for the near future is therefore very unclear.
Meanwhile acidification continues to reduce lake buffer capacity also in calcareous areas, where there is yet no dialogue about damages caused by acid deposition. Due to unique time series from the 1940's, it is possible to demonstrate the development in the limnologically well-known Lake Erken. pH in this lake is still about 8, but the alkalinity has decreased since the mid 1960's. About 25% of the lake's buffer capacity has been lost during the last 20 years (Fig.1), and with unchanged conditions Lake Erken can become acidified within 50 years. If this happens, it will then be easy to count the number of non-acidified lakes in Sweden!
Lake chemical responses to reductions in acid depositions seem to be

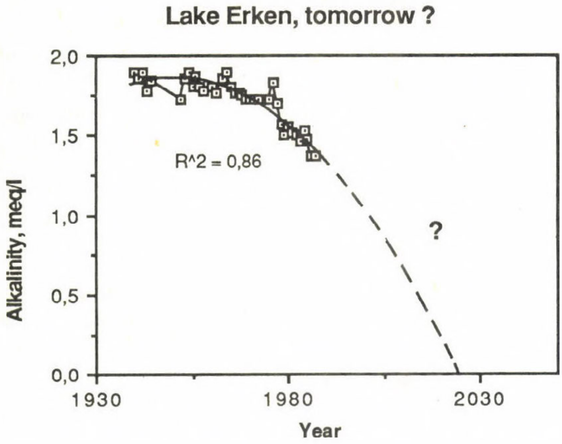


Fig. 1. Alkalinity in Lake Erken, 1940-1987.
Yearly mean values (from 1967, n mostly >11)

hysteretic (Forsberg et al. 1985). This can be expected and explained at the processes level. Acidification is mainly controlled by cation exchange and sulfate retention while deacidification largely depends on silicate weathering (Cosby 1987). With European emission trends in mind, Hauhs (1987) outlines two scenarios of special interest. The first covers acid-sensitive southern Scandinavia with low to moderate acid deposition. After decreasing input, sulfate levels in stream water will respond with improved pH. The response may be delayed by years to decades due to low silicate weathering rates and the necessary change in base saturation. The other scenario refers to heavily loaded, unsensitive and deeply weathered areas in central Europe, where base saturation is declining and equilibrium with current deposition has not yet been established. Here historical depositions may interfere with current and future deposition resulting in an acidification which in practice will be irreversible.

In addition to the already acidified waters, a huge number of lakes situated in different parts of the world are classified as susceptible to acidification. This means that they will become acidic if emissions continue to be above critical levels. The difficulties in reaching international agreements, even on small reductions of 30%, show that it will take time to improve the situation. Unfortunately, it can therefore be concluded, that the number of acidified lakes will continue to increase, at least during this century.

References

Ahl, T.: 1988, Humustransport - tidstrender i rinnande vatten (submitted).

Alcamo, J., Amann, M., Hettelingh, J.-P., Holmberg, M., Hordijk, L., Kämäri, J., Kauppi, L., Kauppi, P., Kornai, G & Mäkelä, A.: 1987, Acidification in Europe: A simulation model for evaluating control strategies. - AMBIO 16:232-245.

Andersoon, F. & Olsson, B. (ed.): 1985, Lake Gårdsjön, an acid forest lake and its catchment. - Ecol. Bull. 37. 336 pp.

Barnaby, F.: 1988, Acid rain: UK policies. - AMBIO 17:160-162.

Barth, H. (ed.): 1987, Reversibility of acidification. Elsevier Applied Science, London, 175 pp.Battarbee, R. W. & Renberg, I.: 1988, Acid deposition and surface acidification: the use of lake sediments. - In Mathy, P. (ed.) Air Pollution and Ecosystems. Proc. Int. Symp. Grenoble, France, May 1987. pp 379-395.

Brakke, D. F., Landers, D. H. & Eilers, J. M.:1988, Chemical and physical characteristics of lakes in the northeastern United States. - Environ. Sci. Technol. 22:155-163.

Cosby, B. J.: 1987, Modelling reversibility of acidification with mathematical models. - In Barth, H. (ed.) Reversibility of acidification. Elsevier Applied Science, London, pp 114-125.Dickson, W. : 1986, Acidification effects in the aquatic environment. - In Schneider, T. (ed.), Acidification and its policy implications, Proc.Int.Conf. Amsterdam. Elsevier, Amsterdam 1986. pp 19-28.

Dickson, W. : 1987, Practical preventive and curative measures for aquatic ecosystems. - In Mathy, P. (ed.) Air Pollution and Ecosystems. Proc. Int. Symp. Grenoble, France, May 1987. pp 469-475.

Dickson, W. : 1988, Liming of Lake Gårdsjön, an cidified lake in SW Sweden. - National Swedish Environmental Protection Board. Report 3426, 327 pp.

Dillon, P. J., Reid, R. A. & Girard, G.: 1986, Changes in the chemistry of lakes near Subury, Ontario following reductions of SO2 emissions. - Water, Air and Soil Pollution 31:59-65.

Driscoll,C. T. & Schafran, G. C.: 1984, Short-term changes in the base neutralizing capacity of an acid Adirondack lake, New York. - Nature 310:308-310.

ECE, Economic Comission for Europe: 1984, Air-borne sulphur pollution. Effects and control. - United Nations, New York, 265 pp

ECE, Economic Comission for Europe: 1987a, Effects and control of transboundary air pollution. - United Nations, New York, 56 pp

ECE, Economic Comission for Europe: 1987b, National strategies and policies for air pollution abatement. - United Nations, New York, 56 pp.

Eilers, M. J., Brakke, D. F. & Landers, D. H.:1988, Chemical and physical characteristics of lakes in the upper Midwest, United States. - Environ. Sci. Technol. 22:164-171.

Forsberg, C.: 1988, Accelerated transport of humic material to Swedish surface waters (submitted).

Forsberg, C. & Löfgren, S.:1988, Dalälvens vatten, 1965-1986. - Länsstyrelsen i Kopparbergs län, Naturvårdsenheten, N 1988:2. 50 pp. (in Swedish).

Forsberg, C. & Morling, G.:1988, Examples of changes in water chemistry during lake acidi- and "deacidification". - Verh. Internat. Verein. Limnol. 23:193-199.

Forsberg, C., Morling, G. & Wetzel, R. G. :1985, Indications of the capacity for rapid reversibility of lake acidification. - AMBIO 14:164-166.

Hauhs, M.:1988, Reversibility of acidification. - In Mathy, P. (ed.) Air Pollution and Ecosystems. Proc. Int. Symp. Grenoble, France, May 1987. pp 407-417..

Hedin, L. O., Likens, G. E. & Bormann F. H.: 1987, Decrease in precipitation acidity resulting from decreased SO_4 concentration- Nature 325:244-246.

Henriksen, A., Dickson, W. & Brakke, D.F. :1986, Estimates of critical loads for sulphur to surface waters. - In Nilsson, J. (ed.) Critical loads for sulphur and nitrogen.The Nordic Council of Ministers, 1986:11, Stockholm. pp 87-120.

Henriksen, A. & Brakke, D. F. :1988, Sulfate deposition to surface waters. Estimating critical loads for Norway and the eastern United States. - Environ. Sci. Technol. 22:8-14.

Kämäri, J.: 1986, Critical deposition limits for surface waters assessed by a process-oriented model. - In Nilsson, J. (ed.): 1986, Critical loads for nitrogen and sulphur. - The Nordic Council of Ministers, 1986:11, pp 121-142.

Landers, D. H., Overton,W. S., Linthurst, R. A. & Brakke, D. F.:1988, Eastern lake survey. - Environ. Sci. Technol. 22:128-135.

Monitor: 1986, Acid and acidified waters. The National Swedish Environment Protection Board. 180 pp (in Swedish).

Neary, B. P. & Dillon, P. J.: 1988, Effects of sulphur deposition on lake-water chemistry in Ontario, Canada. - Nature 333:340-343.

Nilsson, J. (ed.): 1986, Critical loads for nitrogen and sulphur. - The Nordic Council of Ministers, 1986:11, 232 pp.

OECD: 1977, The OECD programme on long range transport of air pollutants; Measurements and Findings. Organisation for Economic Co-operation and development, Paris.

Renberg, I. & Hedberg, T.: 1982, The pH history of lakes in SW Sweden as calculated from the subdossil diatom flora of the sediments. - AMBIO 11: 30-33

Schindler, D. W. : 1987, Recovery of Canadian lakes from acidification. - In Barth, H. (ed.), Reversibility of acidification. Elsevier Applied Science, London, pp 2-13.

Schindler, D. W. :1988, Affects of acid rain on freshwater ecosystems. - Science 239: 149-156.

Schuurkes, J.A.A.R.:1987, Exposure of small-scale aquatic ecosystems to various deposition levels of ammonium, sulphate and acid rain. - In Barth, H. (ed.) Reversibility of acidification. Elsevier Applied Science, London, pp 34-45..

Sullivan, T. J., Eilers, J. M., Church, M. R., Blick, D.J, Eshleman, K.N., Landers, D.H. & DeHaan, M. S.: 1988, Atmospheric wet sulphate deposition and lakewater chemistry. - Nature 331:607-609.

Vangenechten, J. H. D.: 1983, Acidification in West-European lakes and physiological adaption to acid stress in natural inhabitants of acid lakes. - Water Qual. Bull. 8:150-155.

Wright, R. F. & Henriksen, A.:1978, Chemistry of small Norwegian lakes, eith special reference to acid precipitation. - Limnol. Oceanogr. 23:487-498.

Wright, R. F.: 1983, Acidification of freshwater in Europe. - Water Qual. Bull. 8:137-142

Wright, R. F.: 1987, RAIN project: Results after 2 years of treatment. - In Barth, H. (ed.) Reversibility of acidification. Elsevier Apllied Science, London, pp 14-29.

Wright, R. F., Lotse, E. & Semb, A.: 1988. Reversibility of acidification shown by whole-catchment experiments. - Nature 334:670-675.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

PARTIAL RECOVERY OF THE ZOOPLANKTON COMMUNITY IN A SMALL PRECAMBRIAN SHIELD LAKE AS EXPERIMENTAL ACIDIFICATION IS REDUCED

P.S.S. CHANG, D.F. MALLEY

Department of Fisheries and Oceans, Central and Arctic Region,
Freshwater Institute, 501 University Crescent,
Winnipeg, Manitoba, Canada R3T 2N6

INTRODUCTION

Much is known about the deleterious impact of acidification on aquatic ecosystems including zooplankton (Sprules 1975; Keller and Pitblado 1984; Schindler et al. 1985), but there are very few studies on biological recovery from acidification. There has been some documentation of recovery of lakes and streams around Sudbury, Ontario as sulphur emissions have been reduced (Keller and Pitblado 1986; Hutchinson and Havas 1986; MacIsaac et al. 1986). Furthermore, the reduction of U.S. and Canadian sulphur dioxide emissions has allowed some recovery of freshwater systems in Maritime Canada (Thompson 1986). However, there is still much more to learn about the complex process of recovery of aquatic systems from acidification. The purpose of this paper is to provide information on the response of the zooplankton community to the increase in pH from 5.0 to 5.5 in Lake 223, Experimental Lakes Area, Ontario, Canada.

MATERIALS AND METHODS

Lake 223 (49°42'N, 93°43'W; surface area 27.4 ha, maximum depth 14.4 m) is a small Precambrian Shield lake within the Experimental Lakes Area (ELA) in northwestern Ontario. It was experimentally acidified with sulfuric acid from 1976 (initial average pH 6.49) to 1981 (average pH 5.02) and then maintained at pH 5.02 to 5.13 from 1981 to 1983. From 1984 to 1987, acid addition was reduced to allow the pH to rise to 5.5. Details of acidification are given by Cruikshank (1984, 1986).

Zooplankton were sampled using methods as described by Chang et al. (1981) and Chang and Malley (1987). Samples were taken at the deepest point of the lake at monthly or biweekly intervals. Data are presented here for the ice-free season for Lake 223 from 1974 to 1987 (except for 1975 and 1976) and for the nearby reference Lake 239 (56.1 ha in surface area; 30.4 in maximum depth; and circumneutral in pH) from 1970 to 1986 (except for 1973 to 1979). Conversion from abundance to biomass was made using dry weight estimates and length-weight relationships given by Lawrence et al (1987).

RESULTS

The zooplankton community composition in the reference Lake 239 was relatively stable during the early 1970's and the 1980's (Table 1). There was no trend toward reduction in the number of species in any taxonomic group nor major shifts in species. Prior to acidification in 1974, Lake 223 resembled Lake 239 in species composition (Table 2).

Table 1. Occurrence of species in Lake 239. Solid lines represent abundant populations. Dotted lines denote the species were rare. Numbers of rare species are given in parenthesis.

Species Years	1970	1971	1972	1980	1981	1982	1983	1984	1985	1986
Cladocera										
Daphnia galeata mendotae	—	—	—	—	—	—	—	—	—	—
Bosmina longirostris	—	—	—	—	—	—	—	—	—	—
Diaphanosoma birgei	—	—	—	—	—	—	—	—	—	—
Holopedium gibberum	—	—	—	—	—	—	—	—	—	—
Chydorus sphaericus type	—	—	—	—	—	—	—	—	—	—
No. species	5	5	5	5	5	5	5	5	5	5
Calanoida										
Diaptomus minutus	—	—	—	—	—	—	—	—	—	—
Epischura lacustris	—	—	—	—	—	—	—	—	—	—
No. species	2	2	2	2	2	2	2	2	2	2
Cyclopoida										
Cyclops bicuspidatus thomasi	—	—	—	—	—	—	—	—	—	—
Cyclops vernalis				- -	- -	- -	- -			
Mesocyclops edax	—	—	—	—	—	—		—	- -	- -
Tropocyclops prasinus mexicanus	—	—	—	—	—	—	—	—	—	—
No. species	3	3	3	3(1)	3(1)	3(1)	2(1)	3	2(1)	2(1)
Rotifera										
No. species	19	20	19	18	18	25	21	25	19	20

However, Lake 223 lost species and/or fluctuated in biomass in all four taxomonic groups as acidification progressed (Table 2; Fig. 1). The greatest effects of acidification on zooplankton in Lake 223 were the marked increase in cladoceran and rotifer biomass, the decrease in copepod abundance and the shifts of cladoceran species composition.

As experimental acidification was reduced and Lake 223 pH was allowed to recover from 5.0 to 5.5, biomass of cladocerans and rotifers steadily decreased and that of calanoids increased to levels similar to those prior to and in the early acidification years (Fig. 1). Cladoceran species composition showed partial return to the community present in the early stages of acidification. A species of cyclopoid also reappeared (Table 2). There appears to be a drop in species richness of rotifers in 1987.

Table 2. Occurrence of species in Lake 223. Solid lines represent abundant populations. Dotted lines denote the species were rare. Numbers of rare species are given in parenthesis.

Species Years	1974	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987
Mean ice-free season pH	6.64	6.13	5.93	5.64	5.59	5.02	5.09	5.13	5.44	5.53	5.55	5.52
Cladocera												
Daphnia galeata mendotae	———	———	———	———	———	- - -						- - -
Daphnia dubia					———	- - -		- - -			- - -	———
Daphnia catawba					———	———	———	———	———	- - -	- - -	- - -
Bosmina longirostris	———	———	———	———	———	———	———	———	———	———	———	———
Diaphanosoma birgei	———	———	———	———	———	- - -		- - -		- - -	———	———
Holopedium gibberum	———	———	———	———	———	———	———	———	———	———	———	———
Chydorus sphaericus type						- - -					- - -	———
No. species	4	4	4	4	6	3(4)	3	3(2)	3	2(2)	3(3)	5(2)
Calanoida												
Diaptomus minutus	———	———	———	———	———	———	———	———	———	———	———	———
Diaptomus sicilis	———	———										
Epischura lacustris	———	———	———	- - -	- - -							
No. species	3	3	2	1(1)	1(1)	1	1	1	1	1	1	1
Cyclopoida												
Cyclops bicuspidatus thomasi	———	———	———	———	———	———	———	———	———	———	———	———
Cyclops vernalis	- - -	- - -			- - -							
Mesocyclops edax	———	———	———	———	———	———	———	———	———	———	———	———
Tropocyclops prasinus mexicanus	———	———	———	———	- - -	- - -						- - -
Eucyclops speratus									———			
Orthocyclops modestus											- - -	
No. species	3(1)	3(1)	3	3	2(2)	2(1)	2	2	3	2	2(1)	2(1)
Rotifera												
No. species	20	20	22	21	19	21	20	17	20	20	16	13

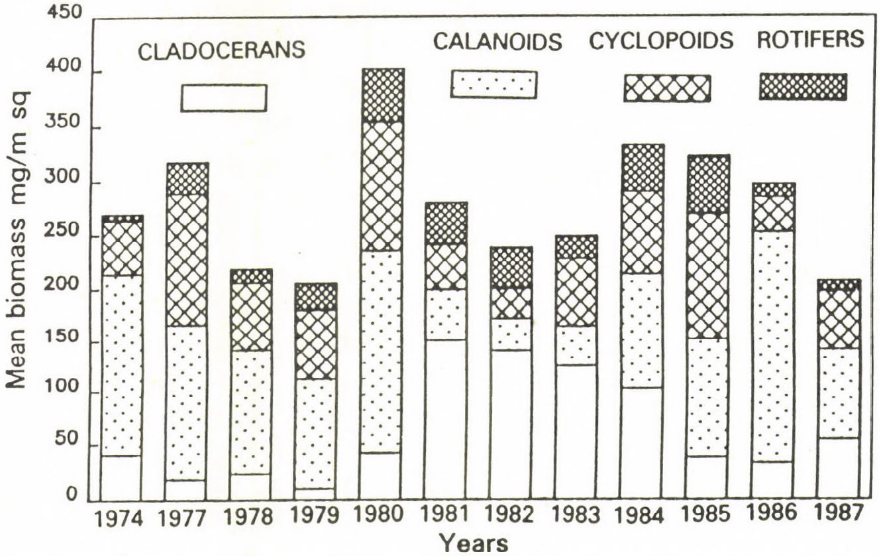


Fig. 1. Zooplankton biomass in Lake 223. Values are open-water means from 1974 to 1987.

DISCUSSION

Schindler et al. (1985) reported acidification of Lake 223 to pH 5.0 eliminated several key species of fishes (e.g., fathead minnows, fry of lake trout and of white sucker) and invertebrates (e.g., Mysis relicta, and crayfish). Some of these are key predators on zooplankters. Correlated with the decrease predation pressure from fish, there was an increase in abundance of Cladocera (Malley and Chang 1986).

Recovery of Lake 223 to a pH of 5.5 by reducing inputs of sulfuric acid allowed three species of small fish to reappear in the lake: pearl dace, white sucker and a newcomer, the three-spined stickleback (Ken Mills, pers. comm., Freshwater Institute, Winnipeg, Man.). Pearl dace and stickleback were abundant during the recovery years, and they are zooplankton feeders. Although the white sucker mainly feeds on bottom organisms, it may have turned into an opportunistic feeder since the overproduction of the population during the recovery period exhausted its food supply. Therefore, since Mysis has not returned to the system and the other major zooplankton predator, Chaoborus, was rare in Lake 223, it is hypothesized that the abundance of these fish species caused the decrease in biomass of cladoceran and rotifers during the recovery from acidification in Lake 223.

In general, biological recovery as exemplified in the zooplankton community, has not followed chemical recovery in the rate of change and is variable in extent among groups of zooplankton. However, the most significant point is that the zooplankton community stressed by acidification showed partial recovery as acid stress was reduced.

REFERENCES

Chang, P.S.S., D.F. Malley, I.L. Delbaere, and G. Mueller. 1981. Species composition and seasonal abundance of zooplankton in Lake 223, Experimental Lakes Area, northwestern Ontario: Before and during acidification, 1974-1979. Can. Data Rep. Fish. Aquat. Sci. 290: iv + 42 p.

Chang, P.S.S. and D.F. Malley. 1987. Zooplankton in Lake 223, Experimental Lakes Area, northwestern Ontario, 1974-1983. Can. Data Rep. Fish. Aquat. Sci. 665: iv + 235 p.

Cruikshank, D.R. 1984. Whole lake chemical additions in the Experimental Lakes Area. 1969-1983. Can. Data Rep. Fish. Aquat. Sci. 449: iv + 23 p.

Cruikshank, D.R. 1986. Whole lake chemical additions in the Experimental Lakes Area. 1984-1985. Can. Data Rep. Fish. Aquat. Sci. 580: iv + 10 p.

Hutchinson, T.C. and M. Havas. 1986. Recovery of previously acidified lakes near Coniston, Canada following reductions in atmospheric sulphur and metal emissions. Water Air Soil Pollut. 28: 319-333.

Keller, W., and J.R. Pitblado. 1984. Crustacean plankton in northeastern Ontario lakes subjected to acidic deposition. Water Air Soil Pollut. 23: 271-291.

Keller, W., and J.R. Pitblado. 1986. Water quality changes in Sudbury area lakes: A comparison of synoptic survey in 1974-1976 and 1981-1983. Water Air Soil Pollut. 29: 285-296.

Lawrence, S.G., D.F. Malley, W.J. Findlay, M.A. MacIver, and I.L. Delbaere. 1987. Method for estimating dry weight of freshwater planktonic crustaceans from measures of length and shape. Can. Fish. Aquat. Sci. 44 (Suppl. 1): 264-274.

MacIsaac, H.J., W. Keller, and N.D. Yan. 1986. Natural changes in the planktonic rotifera of a small acid lake near Sudbury, Ontario following water quality improvements. Water Air Soil Pollut. 31: 791-797.

Malley, D.F., and P.S.S. Chang. 1986. Increase in the abundance of Cladocera at pH 5.1 in experimentally-acidified Lake 223, Experimental Lakes Area, Ontario. Water Air Soil Pollut. 30: 629-638.

Schindler, D.W., K.H. Mills, D.F. Malley, D.L. Findlay, J.A. Shearer, I.J. Davies, M.A. Turner, G.A. Linsey, and D.R. Cruikshank. 1985. Long-term ecosystem stress: the effects of experimental acidification on a small lake. Science (Wash., DC) 228: 1395-1401.

Sprules, W.G. 1975. Midsummer crustacean zooplankton communities in acid-stressed lakes. J. Fish. Res. Board Can. 32: 389-395.

Thompson, M.E. 1986. The cation denudation rate model - its continued validity. Water Air Soil Pollut. 31: 17-26.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

THE IMPORTANCE OF SELF-OXIDATION IN MATTER-ENERGY FLOW IN LAKE ECOSYSTEMS AND ITS pH DEPENDENCE

L.G.-TÓTH, J. ZLINSZKY

Balaton Limnological Research Institute of the Hungarian Academy of Sciences
H-8237 Tihany, Hungary

THE HYPOTHESIS

Decomposition of organic matter depends on the pH of the lake water. Metabolism of well aerated alkaline lakes is rapid and there is practically no accumulation of organic matter. By contrast, in acidified lakes dead organic matter accumulates and the degree of accumulation increases with acidity. In extremely acidic Sphagnum bogs (pH < 4) even the structures of tissues of dead plants and animals have been conserved for thousand years.

Traaen (1980) experimentally proved the negative effect of acidic conditions on the decomposition and this phenomenon has been utilized for food preservation since early cultures of man.

Respiratory electron transport system (ETS) localized on the inner membranes of mitochondria makes a bridge between oxidizing organic compounds and molecular oxygen. The activity of this complex of enzymes in vitro rapidly increases with the increase in pH of medium (Kenner and Ahmed 1975, Owens and King 1975). In our opinion, the pH dependence of decomposition in nature should be connected with the pH dependence of oxidation on a biochemical level.

Oxidation in the tissues of eucaryots can hardly be influenced by the environmental pH, as the inner pH stability is assured by ion pumps. ETS activity of decomposer bacteria can be effected by environmental pH if their cell membranes and diverse cell walls are H-ion permeable, but they also have ion pumps. Presumably their function needs more energy under acidic conditions (Hooper and Dispirito 1985).

As an alternative we propose the hypothesis of pH-dependent self-oxidation which takes place by the postlethal function of ETS. It is probable that pH dependence of biological oxidation increases as soon as the organisms die. Through damages and gaps of cell walls and membranes the internal and external pH equalize. In our opinion, under these circumstances ETS is functioning in dependence of environmental pH, so if the organisms are enough tiny (plankton) it can effectively oxidize itself in alkaline lakes. This process might provide an alternative way of decomposition and directly explains the proved pH dependence of decomposition.

MATERIAL AND METHODS

We wanted to investigate whether ETS activities of organisms deriving from waters with different pH are equally dependent upon pH. In other words, we wanted to test the possibility of enzymatic adaptation. Samples were taken from an acidic Sphagnum bog (Lake Monostori, pH = 3.9-4.2) and from the alkaline Lake Balaton (pH = 8.4-8.6) in Hungary. ETS activity of samples was measured with tetrazolium reducing test at 6 different pH values, respectively. The scheme of experiments is presented in Fig. 1.

Moreover, we wanted to prove that the dead biomass keeps its ETS activity, the oxidation goes on after death and its intensity depends on the external pH. For these experiments (Fig. 2) axenic cultures of the green alga Selenastrum capricornutum Printz in the decaying phase were used. ETS activity, organic carbon content (TOC), pigments and, in order to detect eventual microbial infection, the bacterial DNA synthesis of decaying algal mass were measured in time, in acidic and alkaline conditions.

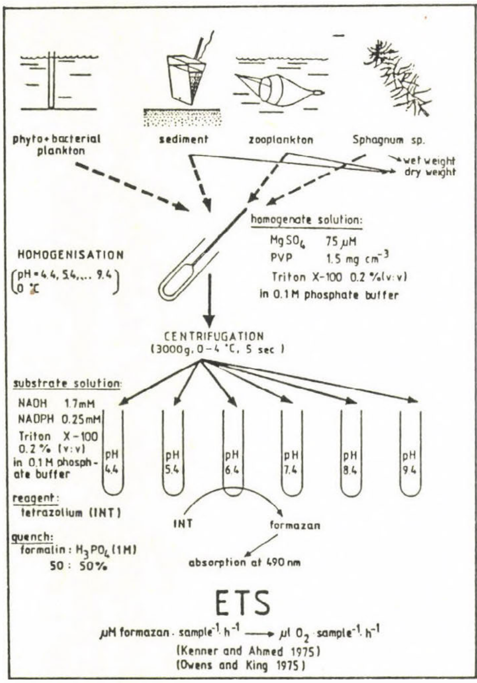


Fig. 1

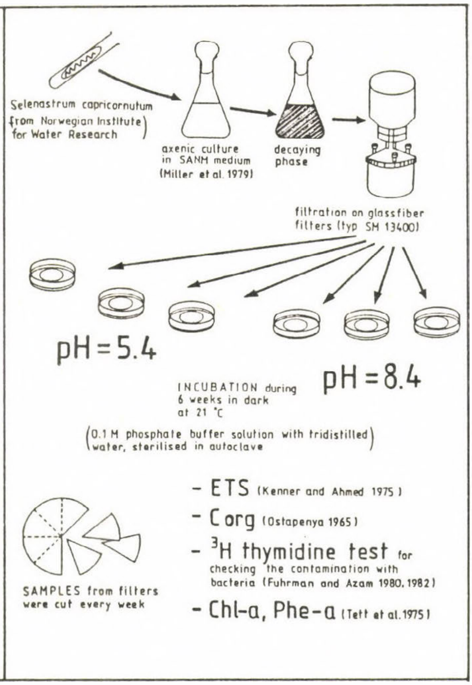


Fig. 2

Fig. 1. Tests to investigate the pH dependence of ETS activity of samples taken in alkaline (Lake Balaton, pH = 8.4-8.6) and in acidic (Sphagnum bog Lake Monostori pH = 3.9-4.1) environments

Fig. 2. Scheme of self-oxidation experiment with axenic culture of the green alga Selenastrum capricornutum Printz

RESULTS

ETS activity of zooplankton, phyto+bacterioplankton, Sphagnum sp. and lake sediment were closely correlated with the pH of the medium. There was no difference in pH dependence of samples derived from alkaline and acidic environments. One unit increase in pH manifested in a 2.6-fold increase in ETS activity between pH = 4.4 and 8.4 (Fig. 3). Above pH = 8.4 ETS activity has saturated.

Selenastrum capricornutum cultured in alkaline medium (decaying phase, dark) had high ETS activity at the beginning of the experiments. Their ETS activity exponentially decreased

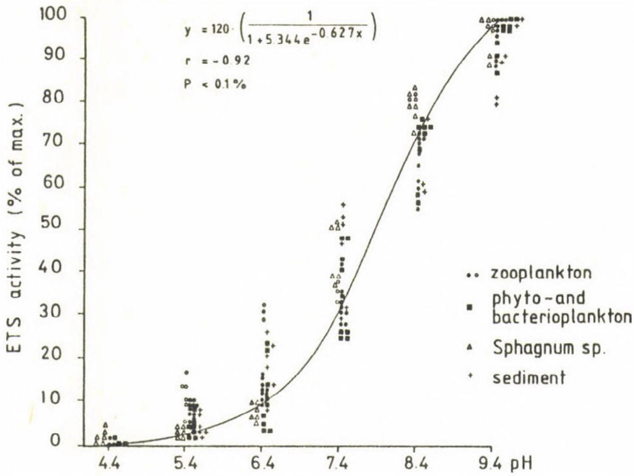


Fig. 3. Effect of pH on ETS activity (% of max.) of homogenates of samples derived from the alkaline Lake Balaton (full symbols) and from acidic Sphagnum bog Lake Monostori (empty symbols)

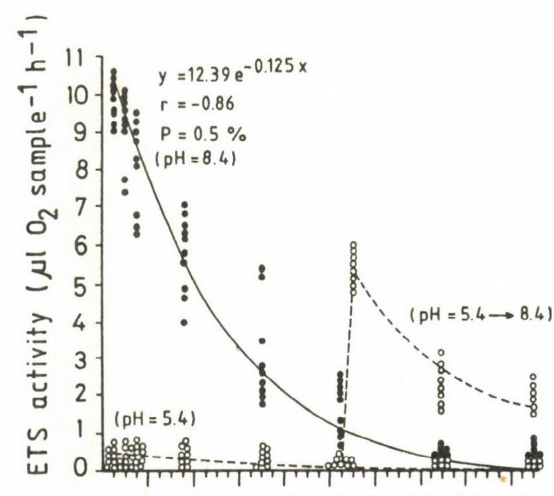


Fig. 4. Changes in ETS activity of axenic culture of Selenastrum in decaying phase kept in alkaline (full circles) and acidic (empty circles) medium (in dark, at 21°C)

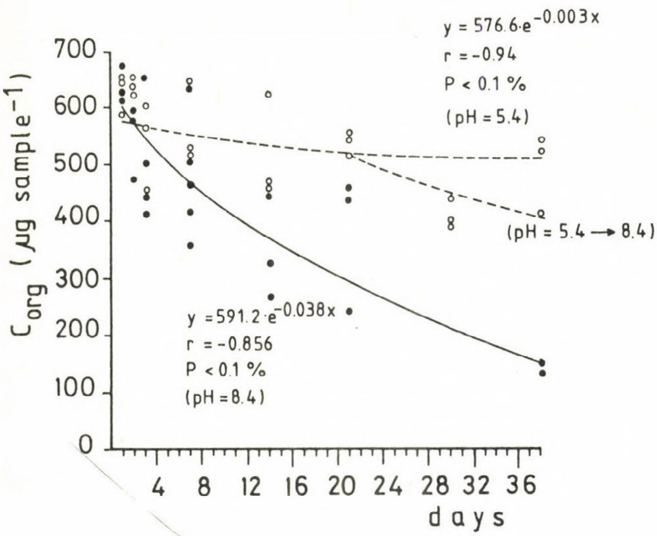


Fig. 5. Changes in organic carbon content of axenic culture of Selenastrum in decaying phase kept in alkaline (full circles) and acidic (empty circles) medium (in dark, at 21°C)

during 6 weeks until the analytical limit of the measurement (Fig. 4). Organic carbon and chlorophyll-a contents of the samples decreased to 19.5% and 17.4%, respectively, of the original values (Fig. 5).

ETS activity of decaying algal mass in acidic medium had been very low during the 6 weeks, their organic carbon content decreased to 85.7% of the original value, then stabilized at about the 3rd to 4th week (Figs 4, 5). Chlorophyll-a content of the samples was 32.0% of the original value at the end of the experiment. When these "acidic" samples were placed in alkaline medium their ETS activity increased.

CONCLUSIONS

Based on the results we can state the following.

1. pH dependence of ETS of organisms deriving from alkaline and acidic environments is similar, thus there exists no enzymatic adaptation.

2. ETS activity of dead biomass can survive and the matter of microscopic organisms self-oxidizes for certain time.

3. Tricarboxylic acid cycle, where the decarboxylation itself happens and which provides the coenzyme supply of ETS, should also function after death.

4. As the ETS and therefore the self-oxidation depend on pH, decomposition in nature also depends on it.

5. The pH of lakes basically determines the efficiency and velocity of nutrient recirculations in lakes. One unit difference in pH between two lakes means 2-3-fold difference in the intensity of decomposition according to the pH dependence of ETS activity. In this way the intensity of decomposition in alkaline lakes (pH > 8) might be even 12 times higher than it is in extremely acidic bogs (pH < 4).

REFERENCES

Fuhrman, J.A. and Azam, F. (1980): Bacterioplankton secondary production estimates for coastal waters of British Columbia, Antarctica and California. Appl. Environ. Microbiol. 39, 1085-1095.

Fuhrman, J.A. and Azam, F. (1982): Thymidine incorporation as a measure of heterotrophic bacterioplankton production in marine surface waters: Evaluation and field results. Mar. Biol. 66, 109-120.

Hooper, A.B. and Dispirito, A.A. (1985): In bacteria which grow on simple reductants, generation of a proton gradient involves extracytoplasmic oxidation of substrate. Microbiol. Reviews 49 (2), 140-157.

Kenner, R.A. and Ahmed, S.I. (1975): Measurements of electron transport activities in marine phytoplankton. Mar. Biol. 33, 119-127.

Ostapenya, A.P. (1965): Method for measuring the particular organic carbon content of water by dichromate. Publ. Russ. Acad. Sci. BSSR 9 (4), 273-276 (in Russian).

Owens, T.G. and King F.D. (1975): The measurement of respiratory electron transport system activity in marine zooplankton. Mar. Biol. 30, 27-36.

Tett, P., Kelly, M.G. and Morberger, G.M. (1975): A method for the spectrophotometric measurement of chlorophyll-a and pheophytin-a in benthic macroalgae. Limnol. Oceanogr. 20 (5), 887-896.

Traaen, T.S. (1980): Effects of acidity on decomposition of organic matter in aquatic environments. Proc. Int. Conf. Ecol. Impact Acid Precip., Norway 1980, SNSF project, 340-341.

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

ANTHROPOGENIC ORGANIC CHEMICAL POLLUTION OF LAKES WITH EMPHASIS ON THE LAURENTIAN GREAT LAKES

R.J. ALLAN

Lakes Research Branch, National Water Research Institute,
Canada Centre for Inland Waters
Burlington, Ontario, Canada L7R 4A6

ABSTRACT

During the last few decades, lakes around the world have become polluted by organic chemicals of anthropogenic origin. Such chemicals enter lakes in industrial discharges, urban effluents, or following their use in agriculture, forestry, and other human activities. These chemicals can be transported long distances to lakes by tributaries or be deposited directly to lakes from the atmosphere. In developed countries of the northern hemisphere, such as the Great Lakes, the major pulse of many organic chemicals, such as PAHs and PCBs to lakes is already past. In developing countries, present-day input of chlorinated organic chemicals to lakes may still be high. The Laurentian Great Lakes are used as the main example for discussion of the evolution of our knowledge in this area, with emphasis on the sources and fate of toxic organic chemicals. In the Great Lakes, detection of chlorinated organic chemicals began in the early seventies and accelerated into the early eighties with sequential discoveries of different compounds. The phase of maximum chlorinated organic chemical pollution of the Great Lakes occurred in the 1960's and early seventies. Few new compounds of equal combined toxicity and concentration have been discovered in the Great Lakes in the last five years and a major recovery is already well underway as evidenced by declines in the concentrations of such organic chemicals in lake bottom sediments, fish and fish-eating birds. Nevertheless, recycling of of toxic organic chemicals from contaminated bottom sediments along with their global scale atmospheric cycling prolongs recovery. Rates of recovery will slow and be related more to limnological factors and processes such as trophic state and sedimentation rates and to the recent quantification of the atmosphere as a major and difficult source to control of several organic chemicals now restricted or banned in the Great Lakes Basin. There is a global lack of information on the state of organic chemical pollution of lakes. However, coordinated worldwide studies of organic chemicals in lakes should be considered. The sites should reflect lakes with direct industrial, urban, and agricultural organic chemical inputs and also include lakes at remote sites where atmospheric input of organic chemicals is the only possible or prime source.

INTRODUCTION

Lakes research has developed from individual, classical, limnological investigations done largely in the quest for general knowledge to today's applied, issue driven, multi-disciplinary, team projects. This change began when the eutrophication of formerly oligotrophic lakes became obvious not only to scientists but to fishermen and the recreational public who wanted to swim in lakes with low phytoplankton densities from beaches free of dead fish and algal scums. In the late sixties, Lake Erie was said to be dying and limnologists predicted greater periods of hypolimnetic anoxia. The public were more concerned with murky water and eutrophication thus became the first "great" lakes issue in the great lakes (Figure 1). Debate over the cause of eutrophication focused on the main plant nutrients of carbon, nitrogen, and phosphorus. The role of other trace elements and micronutrients was also debated but the final decision to lower phosphorus concentrations in municipal treatment plant effluents has been proven to be the correct one, at least for the Great Lakes. However, some shallow lakes which were naturally eutrophic have seen phosphorus removal at sewage effluents implemented with marginal success. For some lakes, there is still debate that nitrogen is the limiting element. My purpose in beginning with this first worldwide lake pollution issue is to emphasize that what at first appears obvious to some scientists, eventually becomes a a far more complex situation when costly controls are envisaged. Non-point sources of phosphorus versus point sources had to be quantified to model and predict the effects of controlling each.

Atmospheric vectors had to be considered as had the issues of physical, chemical and biological regeneration of phosphorus from bottom sediments. The bioavailable phosphorus fraction in suspended particulates transported by rivers to lakes and in the suspended particulates and bottom sediments of the lakes themselves had to be quantified. Limnologists had to shift from empirical relationships to the difficult quantification of such processes as internal phosphorus turnover times and regeneration from bottom sediments which still remain open to conjecture. In response to the eutrophication issue, the level of classical limnological research

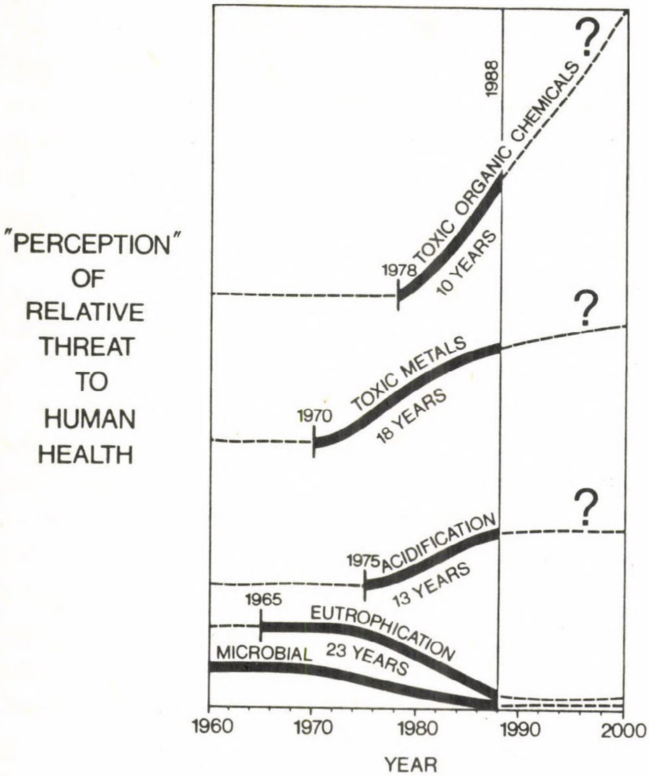


Figure 1. Historical development of issues of lake contamination (in countries with advanced sewage treatment)

expanded greatly in Canada and elsewhere. New institutes were established but their role was no longer simply limnology but rather to the convergence of a variety of scientific disciplines needed to tackle ever new and more complex contamination issues.

The next of these issues was heavy metal pollution (Figure 1) and arose from serious incidents of primarily mercury but also lead pollution. Both these metals are lethal and have serious chronic effects on humans. The heavy metal issue continues today and most contaminant related conferences are still dominated by papers on toxic metal sources, pathways, fate and effects. Today, the effects of toxic metal mixtures are only now being considered when ecosystem objectives are set for lakes. Metals research continues to expand because new analytical techniques detect that soluble metal concentrations are in fact far lower than even recent measurements indicated, now that lake geochemists are beginning to use the clean sampling and laboratory techniques based on developments in the oceanographic sciences. The first international conference directed solely to Trace Metals in Lakes was held only this year. The heavy metals issue, like the eutrophication issue before it, developed from a stage of problem identification to a stage of process research, usually at interfaces such as air-water or sediment-water. Again, many of the processes of metals transfer between lake compartments remain today as relatively crude predictions. Even for mercury, which essentially began the toxic metal issue, studies of methyl-mercury production and cycling in lakes are rare and mainly conducted in the early eighties (Allan et al., 1984). For most laboratories, the routine analyses of large numbers of water, sediment and biological samples for methyl-mercury is still beyond budgetary and analytical capabilities. Today, the ability to investigate a lake-issue, only when the analytical capability makes it possible to do so, is a key point in the status of our present knowledge of anthropogenic organic chemical pollution of lakes.

Soon after the eutrophication and metals issues were well established as lakes research areas, the issue of lake acidification began to gain attention (Figure 1). Research has expanded from around 1975 to the mid-eighties when major international conferences were held. Thus, lakes research has recently, in the last two decades, become more directed by public driven environmental issues such as eutrophication, metal pollution and acidification. All these issues started with problem identification, often relying on new analytical chemical advances and progressed to a multi-disciplinary, team-oriented, process-focused research stage when costly control actions were envisaged based on best available knowledge. The issue of toxic organic chemical pollution of lakes has followed this scenario and we are not yet at the end of the problem identification stage.

Prior to 1980, organic chemical studies in lakes were relatively unsophisticated attempts to identify the problem. Analytical methods, although advanced for the time, were usually not capable of the precision and accuracy accepted in the then nutrients and metals fields. After 1980, when the best funded laboratories began to purchase capillary column and then dual capillary column gas chromatographs, and computerized gas chromatographic-mass spectrometers, the knowledge of the extent of the problem expanded exponentially, at least in the Great Lakes. We are again at a point of transition in the issue to quantification of the processes which control transfer of organic chemicals between lake compartments and lakes and their environment, especially the atmosphere. This need has arisen because of the requirement to complete lake mass balances for critical (high exposure) chemicals to develop organic chemical management plans for lakes.

ANTHROPOGENIC ORGANIC CHEMICAL POLLUTION OF WORLD LAKES

Quite possibly there is now more organic chemical data available on the Great Lakes than for any other lake and the majority of this information has been published in the last ten years. In the recent survey of world lakes (ILEC, 1980) comprehensive summaries of the limnology and pollution status of 66 of the world's largest, most studied, or best-known lakes contains organic chemical data for only four lakes, excluding the five Laurentian Great Lakes. The four lakes are Lake Biwa in Japan for PCBs in sediment and PCBs, DDT, lindane and PCP in fish; Lake Shongkhla in Thailand for DDT and lindane in sediment and DDT in fish; the Zurichzee in Switzerland for TECE and TCE in water; and Lake Victoria in Kenya for DDE in fish.

However, the limited recent literature on many remote lakes supports a general hypothesis that there is or was extensive pollution of world lakes by anthropogenic organic pollutants. (Tanabe _et al_., 1982; Bidleman and Leonard, 1982; Tanabe _et al_., 1983; Norstrom and Muir, 1985; Hoff and Chan, 1986; Villeneuve and Cattani, 1986; Bidleman, 1987.) It is estimated that up to 98% of the PCBs entering the world oceans come from the atmosphere, mainly by gas exchange (Atlas _et al_., 1987), and this may also apply to the world's remote lakes. Pesticides have been detected in fish in remote lakes where their source could only be from the atmosphere. Examples are few but include polychoroterpenes, which are used as rodenticides, in Arctic Char in remote Alpine Lakes (Ballschmiter and Zell, 1980) and DDT and PCBs in Lake Storvindeln in Northern Sweden (Olsson, 1987).

It is only now possible to routinely detect organic chemicals in the low ppt range in 'water'. Thus, there are few historical records of concentrations of such chemicals in water. Even in the Great Lakes, where extensive water quality data has been collected over the last fifteen years, there is no good spatial or temporal data on the concentrations of organic chemicals in "water", with the exception of some volatile organics. Outside the Great Lakes, data is at best scattered. If analyses are performed on "whole" water, the measurements become of somewhat dubious validity because of the extreme partitioning of most toxic chlorinated organic chemicals to suspended particulate matter (Allan, 1986).

Analyses of biotic tissue are presently the most common way of assessing organic chemicals trends in lakes and the most common medium is fish. Many toxic organic chemicals were first detected in the Great Lakes only in the late 1970's and early eighties. Few biological materials had been stored for retrospective analyses with the one key exception of the eggs of the fish-eating, herring gulls.

The second major medium analyzed, especially in the last five years, is lake sediment cores. By dating the cores by analysis for radioisotopes, an historical perspective is added to concentration data. The longest term information on historical trends for anthropogenic organic chemical inputs to world lakes is derived from analyses of lake sediment cores. The concept of using lake sediment cores to detect historic trends in eutrophication and toxic metal pollution was reasonably extensive by the mid-seventies but the common use of cores to detect trends for organic chemicals other than those from combustion sources (PAHs) is a recent development (Durhum and Oliver, 1983; Pavoni et al., 1987).

Analyses of lake sediment cores has been used primarily to demonstrate the global input to lakes of anthropogenic organic chemicals from combustion sources (Figure 2). LaFlamme and Hites (1987) stated that PAHs are found in lake sediments around the world and that the qualitative PAH pattern is consistent for many locations. Although PAHs in remote lakes may be deposited from the atmosphere, Wakeman et al. (1980), cautioned that in three lakes in Switzerland and in Lake Washington in the northwestern United States, the PAHs in sediments may owe their indirect atmospheric origin more to urban runoff. In Lake Washington and in the Greifenzee, Zurichzee and Lake Lucerne, PAH concentrations were relatively uniform until about 1950 (Wakeman et al.,

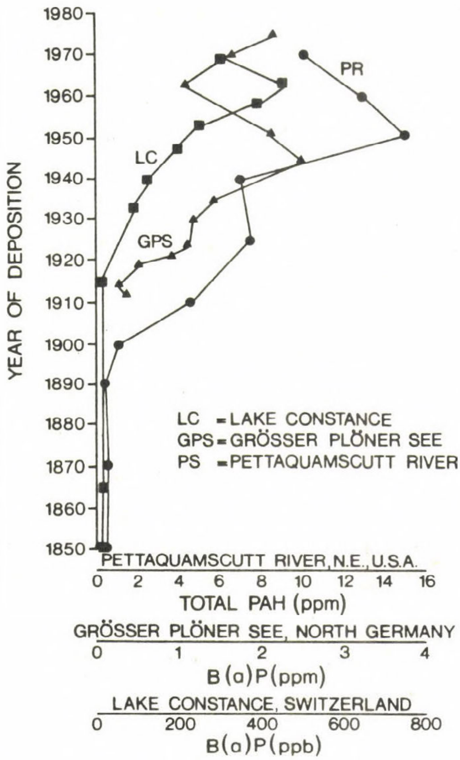


Figure 2. PAHs in lake sediment cores (Adapted from Grimmer and Böhnke, 1975; Muller et al., 1977; and Charles and Hites, 1987)

1980). After 1950, concentrations of PAH have greatly increased. More careful examination of PAHs in cores from remote lakes could allow estimates to be made in other lakes of the direct atmospheric component, a procedure recently used by Swackhammer and Armstrong (1981) to resolve the direct atmospheric deposited fraction of PCBs to Lake Michigan. PAH fluxes to four lakes in Finland were estimated by analyses of datedsediment cores (Wickstrom, et al., 1987) and the PAH mixtures were essentially similar to those in sediment cores from Central-Europe (Grimmer and Böhnke, 1975; Muller et al., 1977) and in both remote and urban lakes in North America (LaFlamme and Hites, 1978; Hites et al., 1980; Gschwend and Hites, 1981; Heit et al, 1981). B(a)P, fluxes of up to 260 $\mu g/m^2/yr$ were determined in the period 1980-84 in Lake Valkeakotinen. In the two most contaminated of these four Finnish lakes, the maximum concentrations were in the surface sediments. The main PAH sources in Finland were considered to be combustion. The analyses of dated sediment cores from a remote area of the Grösser Plöner See showed no significant change from 1915 whereas near built-up areas the concentration of B(a)P was 5 times that of 1915. A similar distribution to this latter B(a)P distribution in the sediment of the Grösser Plöner See was reported by Charles and Hites (1987) for PAHs in a dated sediment core from a site in Rhode Island, in the United States (Figure 2). Peak B(a)P concentrations in the former site and total PAH in the latter both occurred in the mid- to late 1940's. In Lake Constance, Muller et al. (1977) found that the peak B(a)Pconcentrations in a dated sediment core occurred in the early to mid-1960's (Figure 2) and attributed the sources of the PAHs to coal combustion. Gschwend and Hites (1981) found that anthropogenic activities began introducing PAHs to lakes about 80-100 years ago and concluded that PAHs entered lakes from the atmosphere in the particulate phase. Their data indicated a flux of PAHs in the late 1970's of about 5-10 times what it was about 1900 but that the inputs of the 1970's were about 2 times less than they were in the early 1950's.

After the Second World War, the extensive use of DDT and the expanding use of PCBs later added these organic chemicals to the sediments of many lakes. These chemicals generally represent persistent chlorinated pesticides (DDT) and industrial organic chemicals (PCBs). PCBs were first produced commercially in the 1930's and became widely used by the 1960's. Their use in various ways was restricted over the mid- and late seventies but there are still sources including discharges, spills, and incineration. In the oceans, PCB contamination in mussels and other marine bivalves has been detected at many several locations in the Northern Hemisphere, especially around the coasts of North America and Western Europe. There are relatively few data on atmospheric PCB concentrations outside the USA in the northern hemisphere and there is even less data in the southern hemisphere (Tanabe and Tatsukawa, 1987). There is some evidence of a decrease in global atmospheric concentrations of PCBs (Atlas et al., 1987). PCB concentrations in the surface water of the Antarctic, Western Pacific, and Indian Oceans were lower than the values reported for the Atlantic and atmospheric circulation patterns and ocean currents are unfavorable for movement of PCBs between the hemispheres (Richardson et al., 1987). Charles and Hites (1987) determined the PCB maximum to be in the early 1970's in sediment cores from the upper Great Lakes. Between the mid-1970's and the early 1980's, the decrease in input of PCBs to remote northern Wisconsin lakes were estimated at some 30% (Shwackhammer and Armstrong, 1986). In Lake Superior (Eisenreich, 1987) and in Lake Ontario (Durham and Oliver, 1983), the distribution of total PCBs in sediment cores generally correlates with the sales of PCBs.

Most recently, more exotic, highly toxic organic chemicals such as dioxins and furans have been looked for in lake sediment cores. In three lakes in Switzerland (Czuczwa et al., 1985), sediment core profiles indicated that the total PCDD and PCDF concentrations were highest in the immediate surface layers which represented the late 1970's and early 1980's (Figure 3). The initial sharp rise in concentrations of total PCDDs and PCDFs occurred in the late 1940's and early 1950's in the Zurichzee and Baldeggersee. The pre-1940concentrations of these chemical were so low as to be possible of the analytical conditions. The lake sites sampled were all near municipal incinerators and these were considered as a possible source. The bulk of the total dioxins and furans was OCDD. A similar pattern or dioxins was detected in dated sediment cores from Lake Constance, with essentially zero inputs prior to the late 1940's but with a possible peak around the middle 1970's for OCDD (Hagenmaier et al., 1986). Inputs of dioxins and furans to the

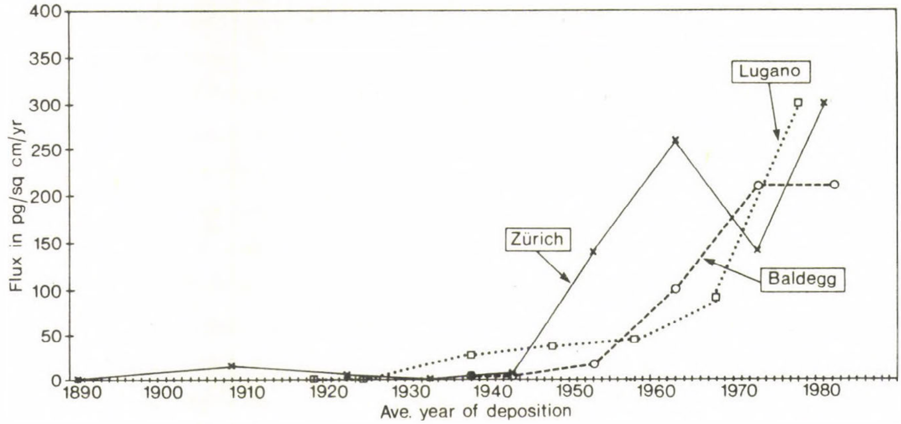


Figure 3. Flux of total PCDD and PCDF to three Swiss lake sediments as a function of time (Czuczwa et al., 1985)

Great Lakes have increased since the 1940's (Czuczwa and Hites, 1984 and 1986). Czuczwa and Hites (1987) proposed that the combustion of coal was not a major source of dioxins to the Great Lakes or elsewhere but that the source was rather the incineration of chemical and municipal wastes high in chlorinated compounds.

The peak inputs of PCDDs and PCDFs to lakes are in the late 1970's or early 1980's and in some cases are highest in the surface sediments. This distribution contrasts with peak lake sediment core PAH concentrations in the 1940's and 1950's and the peaks for PCBs in the mid-1960's to early 1970's. These historical trends appear to reflect three stages of industrial and petrochemical development. The PAHs represent general industrial expansion based on relatively uncontrolled fossil fuel consumption from 1900, with a peak atmospheric input in the 1940-1960 period. The PCBs represent a period of widespread use of organochlorine compounds, including industrial coolants and the early chlorinated pesticides, developed during and soon after World War II and unrestricted until the 1960's and 1970's. The peaks or these compounds occur in the 1960-1975 period and reflect general uncontrolled use and disposal. The declines resulted from health concerns translated into controls for these highly toxic and persistent chlorinated compounds. The dioxins and furans may reflect the very recent reliance on disposal of large quantities of chemical and municipal wastes by incineration (Czuczwa and Hites, 1987).

TRENDS IN ANTHROPOGENIC ORGANIC CHEMICALS IN THE GREAT LAKES

In the Great Lakes, organic chemical concentrations in the aqueous phase are usually in the low ppt range and only recently have large volume samplers been developed so that these levels can be readily detected. Thus, the main sample media used to detect toxic chemical trends in the Great Lakes have been biological, namely predator fish and eggs from herring gull colonies, and lake sediments. In these media, organic chemical concentrations are in the ppm or at least high ppb range. Even so, much of the published data has only appeared since 1980 and especially over the last five years. The analyses of lake sediment cores provides the longest historical trend information because concentrations can be measured in sediment layers deposited before anthropogenic production of the organic chemicals first took place. In Lake Ontario, where most of the sediment core data has so far been obtained, the highs for inputs of toxic organic chemicals to the lakes were in the past (Figure 4). Periods of highest organic chemical concentration in fish and then in herring gull eggs are closely correlated in time but offset from the peaks found in sediment cores, possibly a function of the processes of organic chemical burial in sediments and the time required for the chemicals to bioaccumulate to maximum levels in adult fish and gulls. For all media and all organic chemicals, including 2,3,7,8-TCDD, the general decline in concentrations continues to the present. There are recent fluctuations now that the lowest concentration so far recorded are being measured. The feature of exponential decreases in concentration of persistent toxic organic chemicals in biota has been seen before with toxic metals, for example with Hg introduced to lakes from chlor-alkali plants (Allan _et al._, 1984). In the Hg example, as for come chlorinated organics in the Great

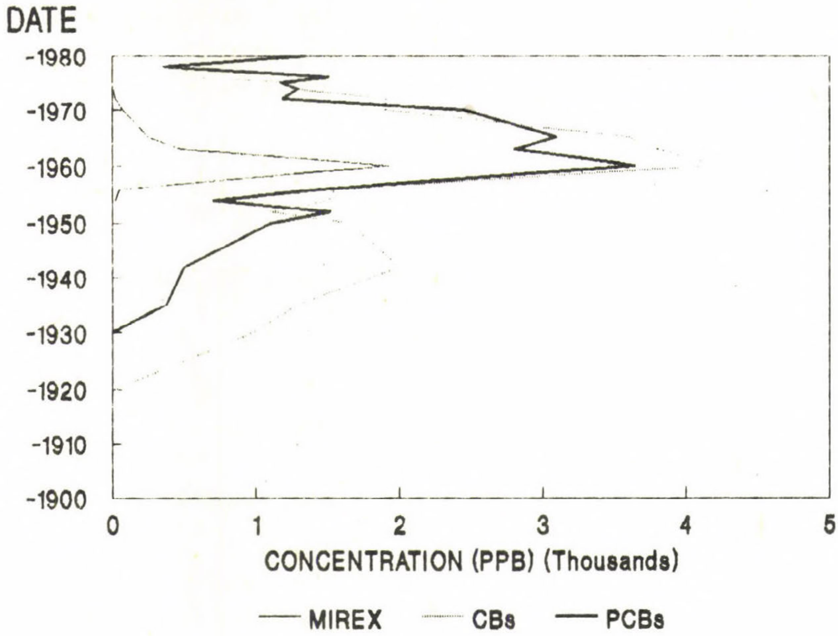


Figure 4. Distribution of organic chemicals in a sediment core from western Lake Ontario (Adapted from Durham and Oliver, 1983)

Lakes, the initial exponential decline in concentration of the persistent/sediment- associated chemical is rapid once the immediate source is removed or dramatically reduced. however, with such compounds, the rate of decline eventually becomes very slow because of internal recycling from historically contaminated sediments and may level out at a concentration that is of concern for some time before all contaminated sediment is buried to a depth below bioturbation activity or below levels affected by physical erosion or resuspension events. A lakewide Nepheloid layer of resuspended bottom sediments was recently discovered in Lake Ontario (Mudroch and Sandilands, 1983). Subsequent analyses of the suspended solids in this layer for chlorinated benzene isomers showed that contaminated surficial bottom sediments are suspended throughout the water column (Oliver and Charlton, 1984). This process of contaminated bottom sediment resuspension along with the accumulation of organic chemicals directly by benthic organisms such as oligochaete worms or chironomids will continue to recycle chemicals into the Lake Ontario food web (Allan, 1986).

Concentration trends for many chlorinated organic chemicals in most predatory fish from the Great Lakes show dramatic historical declines. Concentrations in the early seventies were extremely high for some pesticides such as DDT and some industrial chemicals such as PCBs in some of the lakes, particularly Lakes Michigan and Ontario. Lake Superior and Huron have remained considerably less contaminated, except for specific chemicals in local areas, and Lake Erie concentrations have been lower than might be expected. Trends for DDT in fish are best seen in the long-term data (1970 to 1984/5) for Lakes Michigan and Ontario (Figure 5). The highest DDT concentrations in lake trout from Lake

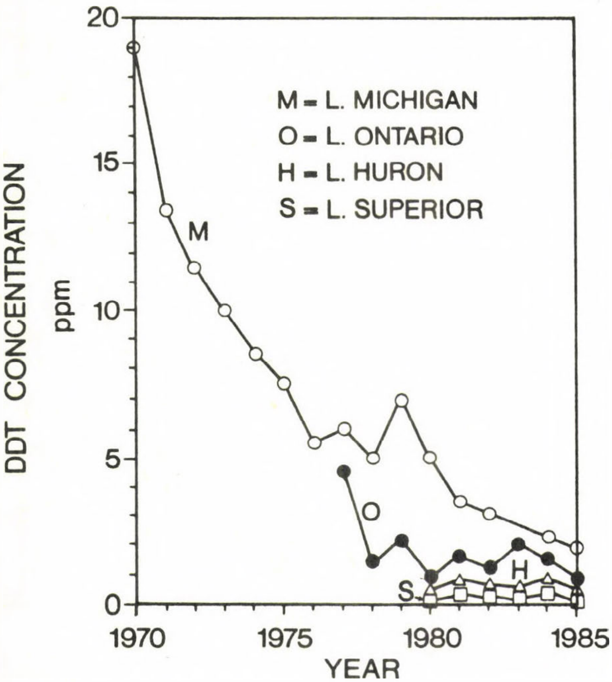


Figure 5. DDT concentration trends in lake trout in the Great Lakes (Adapted from IJC, 1987)

Ontario were around 4.5 ppm in 1977 and by 1985 were reduced to about 1 ppm. Concentrations in Lakes Superior, Erie and even Ontario were much less than in Lake Michigan, where lake trout in 1970 were recorded as containing some 19 ppm DDT but have since declined logarithmically. Other than for Lake Michigan, the trend data for DDT in lake trout begins only after 1977 or 1980 so that it is not possible to make comparisons. Similar declines in concentrations for DDT in lake trout are found for PCBs (Figure 6). In Lake Michigan, a dramatic decline in PCB

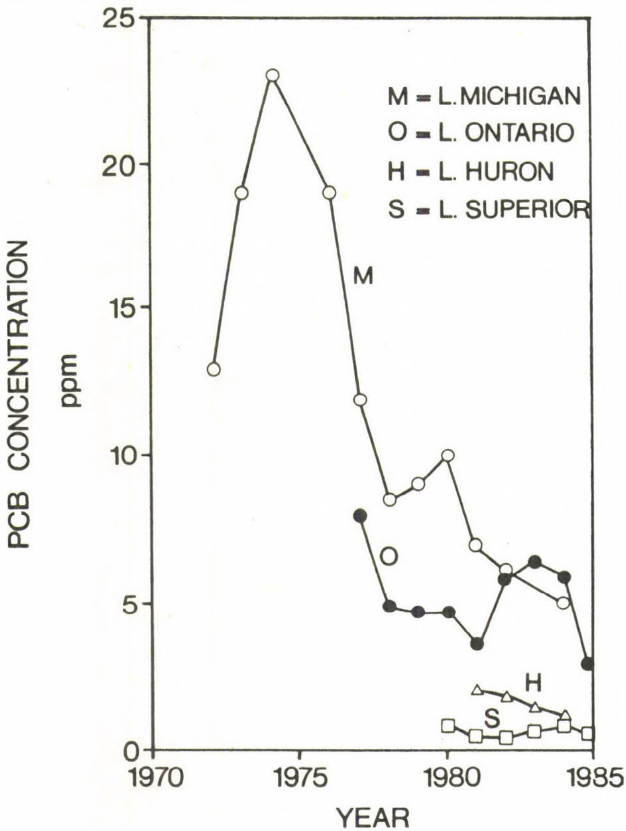


Figure 6. PCB concentrations in lake trout in the Great Lakes (Adapted from IJC, 1987)

concentrations in lake trout took place from a peak of around 24 ppm in 1974 to some 4 ppm by 1984. The 1977 concentrations of PCBs in Lake Ontario lake trout were similar to those in Lake Michigan, namely around 8 ppm. The short period of slight increase in PCB concentrations in fish in Lake Ontario from 1982 to 1984 seems to be a fluctuation at the end of a long decline or related to internal limnological processes.

The eggs of the fish-eating herring gull show similar downward trends in concentrations of chlorinated organic chemicals as those in fish. Gull eggs from colonies on Lake Michigan and Ontario had the highest PCB concentrations (Figure 7) with a possible peak in Lake Ontario in 1974 (Norstrom et al., 1985). Eggs from sites on the other

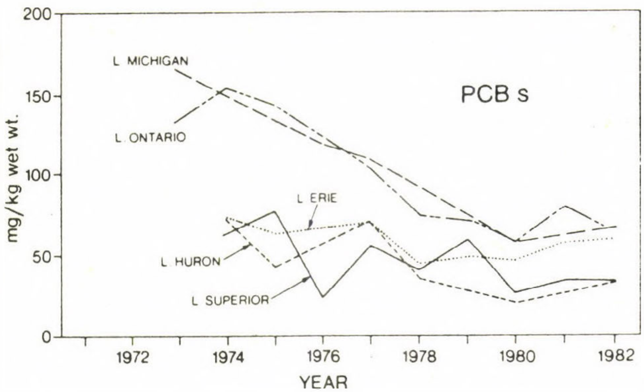


Figure 7. PCBs in herring gulls (Norstrom et al., 1985)

Great Lakes had much lower concentrations of PCBs in the past but the trends with time are all generally downwards. A second example is hexachlorobenzene (Figure 8). The general trend in concentrations for HCB at all colonies is again downward with peak values in 1974, or earlier since no data is available before then. Lake Ontario was the most polluted lake (site) followed by Lake Erie. Both of these lakes have been contaminated by HCB from industrial discharges primarily into the upstream connecting channels of the Niagara and St. Clair Rivers, respectively (Allan et al., 1983; Lawrence, 1986). In 1983, concentration trends for PCBs, and Mirex for Lake Ontario lake trout indicated a leveling out or even an increase in 1982 and 1983. Data now available for 1984 and 1985 (e.g., Figure 6 or PCBs) continue the downward trend. Fluctuations in concentrations may now have as much to do with internal limnological processes such as frequency of storms and

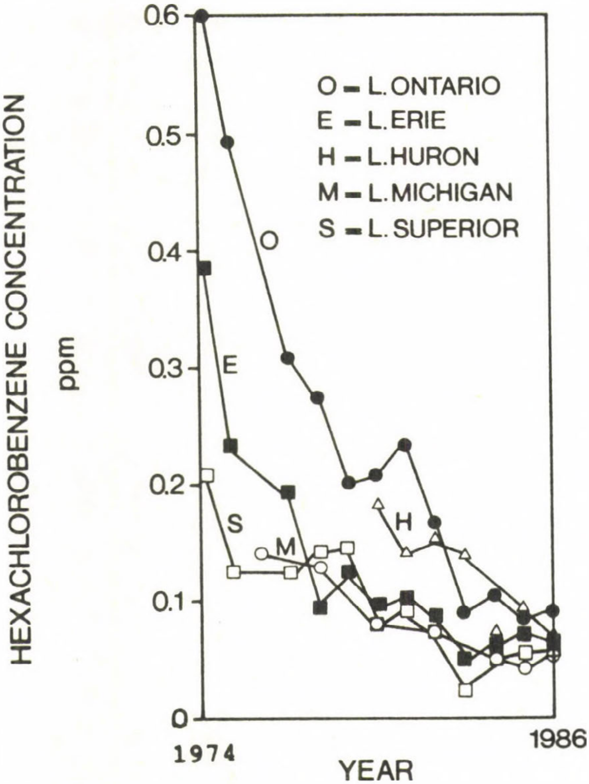


Figure 8. Hexachlorobenzene trends in herring gull eggs from colonies on the Great Lakes (Adapted from IJC, 1987)

contaminated sediment resuspension or annual events of hypolimnetic upwelling as with the actual load of chemicals still entering the lakes, even from the atmosphere in the event of increased precipitation.

THE ATMOSPHERE AS A SOURCE OF ANTHROPOGENIC ORGANIC CHEMICALS TO WORLD LAKES

Many of the most toxic chlorinated organics have been input directly to lakes from point and non-point land-based sources. However, it has become increasingly clear, in the last few years, that a major source for many chlorinated organic chemicals is by deposition from the atmosphere. In both Canada and the United States, the earliest published papers on PCBs in precipitation appeared in the late seventies (Murphy and Rzeszutko, 1977; Swain, 1978; Strachan et al., 1980), although some earlier values existed in reports. Several of the same organic chemicals occur in precipitation in many parts of the world. These chemicals are αBHC, lindane, PCBs, DDT and HCB and they have been recorded, for example, in rain at sites across southern Canada (Strachan, 1988) and in southern France (Villeneuve and Cattini, 1986). From 1980 to present, α-BHC, lindane, dieldrin, pp'-DDE, PCBs and HCB were frequently detected in rain at sites across Canada (Strachan, 1988). The overall mean annual concentrations were 21 ppt for α-BHC and lindane; 3.6 ppt for PCBs; 0.29 ppt for dieldrin; 0.09 ppt for DDE; and 0.12 ppt for HCB. Concentrations of these chemicals in rain samples from across southern Canada and from around the Canadian side of the great lakes were similar, with some minor exceptions. The total organic chemical load to a lake thus depends more on the quantity of rainfall than in variations in organic chemical concentrations in the rainfall itself. These same chemicals have recently been detected in surficial bottom sediments from Great Slave Lake (Figure 9) in Canada's Northwest Territories (Mudroch et al., 1988). Their presence in the Arctic may be due to their deposition from the atmosphere (Norstrom and Muir, 1986). Most of these persistent organic chemicals have been banned or controlled in North America.

Recently, mass balances have been attempted for the main anthropogenic organic chemicals of concern in the Great Lakes (Strachan and Eisenreich, 1987). Although the Great Lakes have possibly more data on organic chemical concentrations in various lake media than any other lakes of the world, the database was extremely poor when the requirement was to close a mass balance equation. Point source and tributary inputs

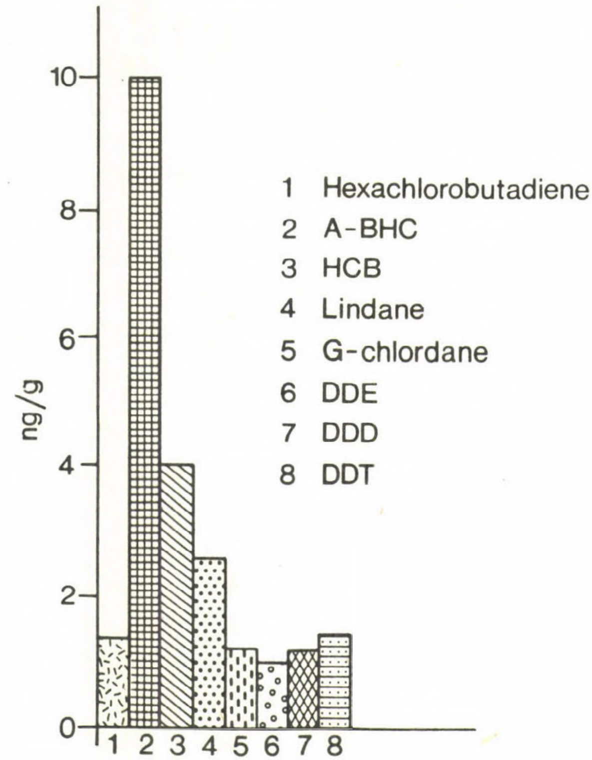


Figure 9. Concentration patterns of selected chlorinated hydrocarbons in surficial sediments from Great Slave Lake (Mudroch et al., 1988)

were usually unquantified as were non- point source inputs from runoff, combined sewers or from groundwater. Analyses of "water" to obtain real dissolved concentrations are few and the use of non-detected values in calculations of loads and lake content became a complex problem. At the end of an International Workshop (Young et al., 1988), PCBs, and perhaps DDT and B(a)P, were the only organic chemicals for which mass balances could be completed with some confidence and atmospheric loads determined. The calcula- tions for PCBs, the most reliable, resulted in estimates that 90% of load to Lake Superior came from the atmosphere. It is interesting to the PCBnote that this value of 90% (606 kg) for a very large (globally) and remote lake begins to approach the estimate for the fraction of the PCBs in world's open oceans (98%) which are thought to come from the atmosphere (Atlas et al, 1987). The atmospheric load of PCBs to Lake Ontario is some 7% of a total of some 2500 kg/yr. The difference is primarily related to the areal dimensions of the two lakes. The atmospheric components of PCB loads to Lakes Michigan, Huron and Erie were calculated at 58%, 78%, and 13%, respectively. For B(a)P, the atmospheric component was 96% for Lake Superior and 72% for Lake Ontario, respectively (Figure 10). It will take great effort and time to refine these rough calculations but will be a likely future requirement as part of lake management plans for organic chemicals.

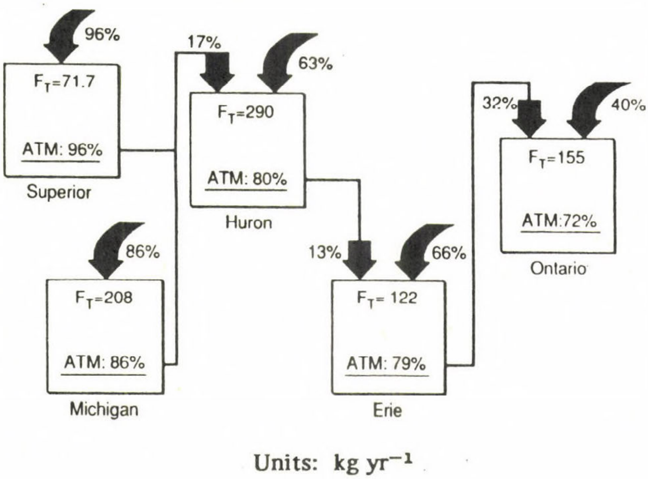


Figure 10. Atmospheric loading of Benz(a)pyrene to the Great Lakes (Strachan and Eisenreich, 1988)

By determining the historical input of organic chemicals to remote lakes, we not only assess their contamination but also estimate historical chemical alterations of the earth's atmosphere. Of equal importance to humans is the fate of such chemicals in remote lakes (and lakes such as the Great Lakes if and when non-atmospheric sources of such chemicals are reduced). Recently, in remote lakes of the southern

Canadian Shield where the only source of organic chemicals could be from the atmosphere, a relationship has been established between PCB levels in zooplankton and the spring total phosphorus levels of the lakes (Carey, 1988). The relationship is presently being refined but implies that lakes of lower trophic state or low spring total phosphorus are lakes where PCBs are bioaccumulated to higher levels in zooplankton (Figure 11). This hypothesis may help to explain the often seemingly random variations in concentrations of PCBs and other toxic organic chemicals in fish from remote lakes. A secondary aspect to this hypothesis but of equal global importance, was an attempt to resolve why PCBs (and

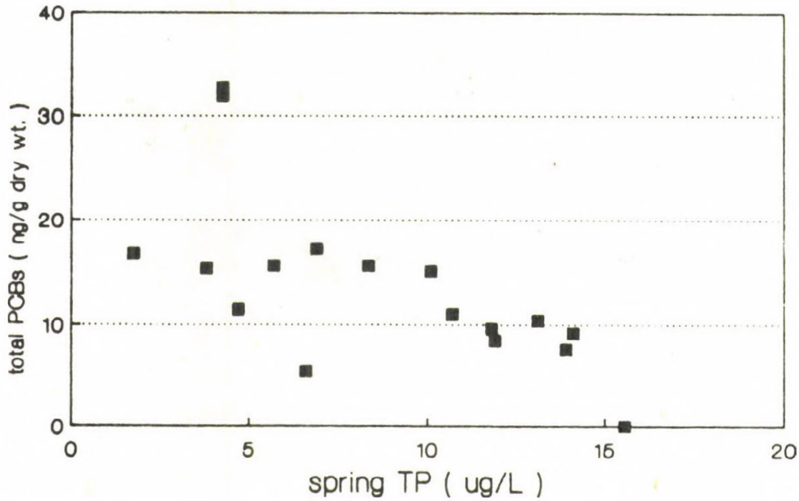


Figure 11. Relationship between spring total phosphorus and zooplankton total PCB concentrations in lakes in Eastern Ontario (Carey et al., 1988)

chemicals of similar properties) appear to be bioaccumulated to lower concentrations in eutrophic lakes versus oligotrophic lakes. The overall loads of PCBs to Lakes Erie and Ontario, given the data available, are almost identical at 2.5 tonnes/year (Strachan and Eisenreich, 1988). It has been speculated that the higher trophic state of Lake Erie which results in an overall faster sedimentation rate, greater competition for organic chemicals by particulates, greater degradation due to higher

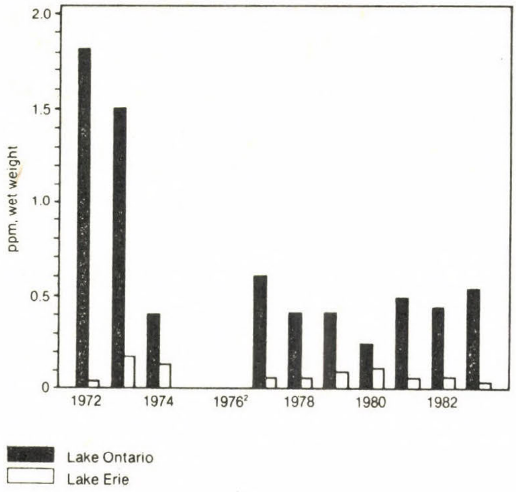


Figure 12. PCB levels in smelt from Lake Ontario and Lake Erie, 1972-1983 (1972-77 data: edible portion; 1977-1983 data: whole fish) (Environment Canada, 1987)

microbial densities, or simply a greater biomass in which to dilute organic chemicals, resulted in lower concentrations of chlorinated organic chemicals in all trophic levels of biota versus those in the other (oligotrophic) Great Lakes (Figure 12). For other world lakes, inputs of organic chemicals from the atmosphere may thus result in concentrations in biota consumed by humans that are not necessarily related simply to atmospheric load, which at most sites on at least a sub-continental scale may we be quite similar, but the limnological processes which control organic chemical bioaccumulation in lakes of different trophic state.

CONCLUSIONS

In general, the extent of global organic chemical pollution of lakes is largely unknown. data exists for some western European lakes based primarily on sediment and biota analyses. For most of the rest of the world, data on organic chemicals in lakes is very rare. When some information does exist, it is usually where fish are a food source, for examples from Lake Victoria or Lake Tiberias. Data on organic chemical fate in remote lakes is extremely rare. The Great Lakes have been relatively intensively studied, although the ability to close mass balances is still poor. Global trends for organic chemical pollution of lakes are known only at a few sites, mainly in North America and western Europe, and mainly for relatively easily analyzed, chlorinated organic chemicals such as of industrial and agricultural origin (PCBs, DDT, lindane, dieldrin) or products of combustion (PAHs) and incineration (PCDD and PCDFs). The atmosphere is a major and rapid dispersion medium, for organic chemicals which are then loaded to lakes directly or indirectly from the drainage basin in the vapor phase or as wet and dry tion. These organic chemicals so introduced to lakes, can deposi- bioaccumulate by various processes to levels that may be of concern even when the aqueous concentrations are below detection limits.

If we are to fill this knowledge gap, then the three prime sample media to be considered would be precipitation, biota and sediments. Even in the Great Lakes basin, wet, dry, and vapor phase precipitation collection devices are in a developmental stage. Precipitation analyses would provide no immediate historical component and such a global network could be prohibitively expensive because of the equipment required and analytical costs at extremely low concentrations. Most results would be beneath detection and trends could not be discerned for many years. Such sampling is only valid in specific, highly-populated, industrial areas where it may be critical to close a lake mass balance for a chemical so as to develop justifiable chemical control scenarios. Analyses of biota is complicated by species differences and age considerations. Also, biological media banks do not exist for most lakes, so there would again be no immediate historical component. Even for the Great Lakes, the peak inputs of several chemicals cannot be seen from retrospective analyses of biological samples because they were often neither archived nor analyzed early enough. Analytical procedures for biological materials are complex, but detection limits are less of a problem than for water. The last medium of lake sediment cores is thus proposed as the tool with which to assess the global extent of organic chemical pollution of lakes and determine historical trends into the past to dates prior to the first appearance of anthropogenically-derived organic chemicals. Analyses and radiodating would be relatively routine at suitably equipped laboratories. Standard samples for quality control are available.

Our knowledge of global organic chemical pollution of lakes could also be greatly extended if data were obtained on toxic organic chemical concentrations in both the sediments and biota of most larger lakes. Nutrient-contaminant interactions should be resolved in lakes where toxic organic chemicals are derived primarily from the atmosphere. The atmospheric component of toxic organic chemical inputs to lakes should be quantified in lakes where such chemicals are also derived from industrial, municipal, and agricultural sources. Finally, lake sediment cores should be collected from appropriate remote lakes, age dated radiochemically, sectioned and analyzed to detect recent the historical trends of atmospheric inputs of toxic organic chemicals. These latter lakes should be headwater lakes, where possible, should stratify and if possible, be meromictic with high sedimentation rates. Global patterns of transport and deposition for specific anthropogenic organic chemicals might emerge. Models have recently been developed to predict atmospheric organic chemical concentrations from analyses of lake sediment core sections (Astle et al., 1987) .

The core group(s) of organic chemicals to be studied would be PAHs, PCBs, HCB, dioxins, lindane, dieldrin, toxaphene and DDT and its metabolites. These analyses are now common at most advanced laboratories and appropriate exchange and development opportunities would permit the transfer of analytical technology to countries around the world. Of course, once the cores were collected and dated, many other analyses (e.g., metals, radionuclides, other organics) could eventually be performed on the archived samples. One main requirement in such a global programme would be participation in a centralized quality control programme which would allow comparison of results around the world. The first phase of the programme could involve some ten countries representing the northern, equatorial and southern areas of the globe. Following an analysis of results, a larger scale programme might be envisaged which would involve locations in all continents and on oceanic islands.

REFERENCES

Allan, R.J., A. Mudroch and M. Munawar. (Eds) 1983. The Niagara River/Lake Ontario Pollution Problem. Special Issue of the J. of Great Lakes Research 9(2):109-340 (231 pp)

Allan, R.J., T. Brydges, D. Dodge, R.D. Hamilton, D.G. Jeffs, and K. Shikaze, 1984. Mercury pollution in the Wabigoon-English River system of northwestern Ontario, and possible remedial measures. Final Report of the Steering Committee, Vol. 1, 18 pp.

Allan, R.J. 1986. The role of particulate matter in the fate of contaminants in aquatic ecosystems. Inland Waters Directorate Scientific Series, No. 142, 128 pp.

Astle, J.W., F.A. Gobas, W-Y. Shiu and D. Mackay. 1987. Lake sediments as historical records of atmospheric contamination by organic chemicals. In: R.A. Hites and S.J. Eisenreich (Eds). Sources and Fates of Aquatic Pollutants. Advances in Chemistry Series 216. Washington, DC: American Chemical Society. pp 57-77.

Atlas, E., T.F. Bidleman and C.S. Giam. 1987. Atmospheric transport of PCB to the oceans. In: J.S. Waid (Ed).PCBs and the Environment. Vol. I. Boca Raton, FL: CRC Press. pp. 79-100.

Ballschmiter, K. and M. Zeal. 1980. Baseline studies of the global pollution. Int. J. Environ. Anal. Chem. 8:15-35.

Bidleman, T.F. and R. Leonard. 1982. Aerial transport of pesticides over the northern Indian Ocean and adjacent seas. Atmos. Environ. 16:1099-1107.

Bidleman, T.F. 1987. Semivolatile organic compounds in the atmosphere: a regional and global perspective. Proc. Tech. Transfer Conference, Toronto, p 55-70.

Carey, J.H., D.R.S. Lean and J.H. Hart. 1988. Processes affecting contaminant distribution and dynamics in lakes and estuaries. Lakes Research Branch, Current Research, 1987-88, p. 77-80.

Charles, M.J. and R.A. Hites. 1987. Sediments as archives of environmental pollution trends. In: R.A. Hites and S.J. Eisenreich (Eds). Sources and fates of Aquatic Pollutants. Advances in Chemistry Series 216. Washington, DC: American Chemical Society, p. 365-389.

Czuczwa, J.M., and R.A. Hites. 1984. Environmental fate of combustion-generated polychlorinated dioxins and furans. Environ. Sci. Tech. 18:444-450.

Czuczwa, J.M. and R.A. Hites. 1986. Airborne dioxins and dibenzofurans: sources and fates. Environ. Sci. Tech. 20(2):195-200.

Czuczwa, J.M. and R.A. Hites. 1986. Sources and fate of PCDD and PCDF. Chemosphere 15(9-12):1417-1420.

Czuczewa, J.M., F. Niessen, and R.A. Hites. 1985. Historical record of polychlorinated dibenzo-p-dioxins and dibenzofurans in Swiss Lakes. Chemosphere 14(9):1175-1179.

Durham, R.W. and B.G. Oliver. 1983. History of Lake Ontario contamination from the Niagara River by sediment radiodating and chlorinated hydrocarbon analysis. J. of Great Lakes Res. 9(2):160-168.

Eisenreich, S.J. 1987. The chemical limnology of nonpolar contaminants: Polychlorinated biphenyls in Lake Superior. In R.A. Hites, and S.J. Eisenreich (Eds). Sources and Fate of Aquatic Pollutants. Advances in Chemistry Series 216. Washington, DC: American Chemical Society. p. 393-469.

Environment Canada. 1987. State of the Environment Report.

Giger and Schaffner. 1977. (in both urban and remote sites in North America, Heite et al., 1981).

Grimmer, G. and H.Böhnke. 1975. Profile analysis of polycyclic aromatic hydrocarbons and metal content in sediment layers of a lake. Cancer Letters, 1:75-84.

Grimmer, G. and H. Böhnke. 1977. Investigation on drilling cores of sediments of Lake Constance: I. Profiles of the polycyclic aromatic hydrocarbons. Z. Naturforsch. 32c:703-711.

Gschwend, P.M. and R.A. Hites. 1981. Fluxes of polycyclic aromatic hydrocarbons to marine and lacustrine sediments in the northeastern United States. Geochim Cosmochim Acta 45:2359-2367.

Hagenmaier, H., H. Brunner, R. Haag, and A. Berchtold. 1986. PCDDs and PCDFs in sewage sludge, river and lake sediments from Southwest Germany. Chemosphere 15(9-12): 1421-1428.

Heit, M., Y. Tan and C. Klusek. 1981. Anthropogenic trace elements and polycyclic aromatic hydrocarbon levels in sediment cores from two lakes in the Adirondack Acid Lake Region. Water, Air and Soil Poll. 15:441-464.

Hites, R.A., R.E. LaFlamme and J.G. Windsor, Jr. 1980. Polycyclic aromatic hydrocarbons in marine/aquatic sediments: Their ubiquity, Chapt. 13 in The Marine Environment. Adv. Chem. Ser. 185:289-311.

Hoff, R.M. and K.W. Chan. 1986. Atmospheric concentrations of chlordane at Mould Bay, NWT, Canada. Chemosphere 15:449

IJC. 1987. Great Lakes Water Quality Board Report, 236 pp.

ILEC. 1987. Data Book of World Lake Environments - A Survey of the State of World Lakes. Interim Report (1), 1987-89. Pub. ILEC/UNESCO,Otsu, Japan

LaFlamme, R.E. and R.A. Hites. 1978. The global distribution of polycyclic aromatic hydrocarbons in recent sediments. Geochim. Cosmochim. Acta 42:289-303.

Lawrence, J. (Ed). 1986. St. Clair River Pollution. Special Issue, Water Poll. Res. Jour. of Canada 21(3):283-459/

Mudroch, A., R.J. Allan and S.R. Joshi. 1988. Preliminary investigation of toxic chemicals in the sediments of Great Slave Lake, Northwest Territories, Canada (Manuscript).

Muller, G., G. Grimmer and H.Böhnke. 1977. Sedimentary record of heavy metals and polycyclic aromatic hydrocarbons in Lake Constance. Naturwissenschaften 64:427-431.

Murphy, T.J. and C.P. Rzeszutko. 1977. Precipitation inputs of PCBs to Lake Michigan. J. Great Lakes Res. 3(3-4):305-312

Norstrom, R.J. and D.C.G. Muir. 1986. Long-range transport of organochlorines in the arctic and Sub-Arctic: Evidence from analysis of marine mammals and fish. In N.W. Schmidtke (Ed) Toxic Contamination in Large Lakes. Volume I: Chronic effects of toxic contaminants in large lakes. Chelsea, MI: Lewis Publishers. p. 83-112.

Oliver,B.G. and M.N. Charlton. 1984. Chlorinated organic contaminants on settling particulates in the Niagara River vicinity of Lake Ontario. Environ. Sci. Technol. 18:903-908.

Olsson, M. 1987. PCBs in the Baltic environment. In J.S. Waid (Ed). PCBs and the Environment. Vol. III. Boca Raton, FL: CRC Press, p. 181-208.

Pavoni, B, A. Sfrisco and A. Marcomini. 1986. Concentration and flux profiles of PCBs and PAHs in a dated sediment core from the Lagoon of Venice. Marine Chemistry 21:25-35.

Richardson, B.J., R.H. Smillie and J.S. Waid. 1987. Case study: The Australian experience. In: J.S. Waid (Eds) PCBs and the Environment. Vol. III. Boca Raton, FL: CRC Press. p. 241-263.

Sandilands, R.G. and A. Mudroch. 1983. Nepheloid layer in Lake Ontario. J. Great Lakes Res. 9(2):190-200.

Strachan, W.M.J. 1988. Atmospheric deposition of organic pollutants. Lake Research Branch, Current Research, 1987-88, p. 122-124.

Strachan, W.M.J. and S.F. Eisenreich. 1988. Mass balancing of toxic chemicals in the Great Lakes: The role of atmospheric deposition. IJC Report, Publ. IJC, Windsor, 113 pp.

Strachan, W.M.J. H. Huneault, W.M. Schertzer, and F.C. Elder, 1980. In: Hydrocarbons and Halogenated Hydrocarbons in the Aquatic Environment. B.K. Afghan and D.Mackay (Eds). Plenum Press, New York, pp. 387-396.

Swackhammer, D.L. and D.E. Armstrong, 1986. Estimation of the atmospheric and non-atmospheric contribution and losses of PCBs for Lake Michigan on the basis of sediment records of remote lakes. Environ. Sci. Tech. 20:879-883.

Swain, W.R. 1978. Chlorinated residues in fish, water and precipitation frm the vicinity of Isle Royale, Lake Superior. J. Great Lakes Res. 4:398-407.

Tanabe, S. and R. Tatsukawa, 1987. Distribution, behavior, and load of PCBs in the oceans. In: J.S. Wade (Ed). PCBs and the Environment, Vol. I. Boca Raton, FL: CRC Press. pp. 143-161.

Tanabe, S., H. Hidaka and R. Tatsukawa. 1983. PCBs and chlorinated hydrocarbon pesticides in the Antarctic atmosphere and hydrosphere. Chemosphere 12:277-288.

Tanabe, S., T. Tatsukawa, M. Kawano and H. Hidaka. 1982. Global distribution and atmospheric transport of chlorinated hydrocarbons: HCH(BHC) Isomers and DDT compounds in the western Pacific, eastern Indian and Antarctic Oceans. J. Oceanog. Soc. Japan 38:137.

Thomas, R.L. and R. Frank. 1987. Introduction in Ecological Effects of in-situ Sediment Contaminants. Hydrobiologia 149:1-4

Villeneuve, J-P. and C. Cattini. 1986. Input of chlorinated hydrocarbons through dry and wet deposition to the western Mediterranean. Chemosphere, 15(2):115-120.

Wakeman, A.G., C. Schaffner and W. Giber. 1980. Polycyclic aromatic hydrocarbons in recent lake sediments: I. Compounds having anthropogenic origins. Geochim Cosmochim Acta 44:403-413.

Wickstrom, K. and K. Tolonen. 1987. The history of airborne polycyclic aromatic hydrocarbons (PAH)and perylene as recorded in dated lake sediments. Water, Air and Soil Poll. 32:155-175.

Young, W.S., P. Wise, R.J. Allan, L. Machta and T. Wagner. 1988. Atmospheric deposition of toxic chemicals to the Great Lakes. Summary Report to the Canada-United States International Joint Commission. Pub. IJC, Windsor, 41 pp.

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

CONTAMINATION OF LIMNIC SYSTEMS BY TOXIC SUBSTANCES AND ITS CONTROL BY BIOMONITORING

G. GUNKEL

Technische Universität Berlin
Strasse des 17. Juni 135, Sekr. KF 8, 1000 Berlin 12

The sources and sites of toxic substance input in limnic systems are often either unknown or unsufficient known. Survey programs are therefore necessary to identify and localize input positions and contaminated areas as well as to develop trend analyses. A developed concept for biomonitoring toxic substances in limnic systems should lead to quantification of toxic effects on the basis of biological reactions, such as bio-uptake of heavy metals. The exact ecological behaviour of a toxic substance in a limnic system must be known in order to decide which compartments of the ecosystem to monitor. These compartments, e.g. abiotic parts or organisms with a significant feeding behaviour and feeding strategy, must be evaluated for their significance in system biomonitoring with regard to bio-uptake, kinetics of bio-uptake, and rank in the hierarchy of the limnic structure.

A concept of biomonitoring heavy metals (Zn, Cu, Cd) with regard to metal speciation in water and residue in macrophytes, organisms and sediments is hereby presented.

1. ECOTOXICOLOGICAL EVALUATION OF TOXIC SUBSTANCES IN LIMNIC SYSTEMS

The significance of a toxic substance in an aquatic ecosystem is determined first by its toxic effects on organisms or populations (acute toxicity), second by its toxic effects on physiological reactions and ecological processes (sublethal effects), and third by the pesticide flux between sediments, water, seston and organisms (transformation processes). Investigations on toxicological effects are necessary and lead to an initial evaluation of the pesticide. It is also necessary to obtain information about the fate of the pesticide in a limnic system and, also, methods are needed to recognize and quantify small amounts of the pesticide in an ecosystem. This knowledge leads to a better evaluation and provides a basis for ecosystem management.

It is therefore imperative to develop methods and strategies for pesticide monitoring. The first step is to take into account all significant abiotic and biotic reations. This implies a classification of the ecosystem-contaminating pesticides into two categories:

1) those characterized by abiotic reactions (e.g. chelating and precipitation) and
2) those characterized by biotransformation processes (e.g. bio-uptake and methylation).

Pesticides in limnic systems can be differentiated into at least five groups of substances with characteristic behaviour. The first two groups are lipophilic and hydrophilic organic substances in a molecular state (e.g. polychlorinated biphenyls and phosphoric acid ester, respectively), which determine the bioaccumulation level in an ecosystem as a steady state reaction (Gunkel 1987). The third group comprises organic substances of a ionic character (e.g. paraquat) that behave a little bit like heavy metals. The fourth group, heavy metals, are characterized by adsorption, ion exchange, and chelate binding (Förstner & Wittmann 1981). The fifth group are metals like mercury, arsenic and tin which undergo methylation, leading to the formation of lipophilic compounds (Mitra 1986). For each of these groups of pesticides, a seperate monitoring program under consideration of the specific reactions of the pesticide must be developed.

The general concept of the present pesticide monitoring program is presented in Fig. 1: The first step is to select one or more monitor organisms or compartments of the ecosystem, such as fish, seston, macrophytes, sediments. Sampling and residue analyses point to the current situation of charge. Using representative sampling points, it is possible to carry out the monitoring, which in turn leads to the determination of trend data - this must, of course, be a feedback system. Intensive sampling during a specific time interval in a localized area also leads to cadastre of pesticide charge and localization of input positions - this, too, must be a feedback system.

Heavy metals are of great significance for ecosystem charge and, in limnic systems, they can contaminate organisms as well

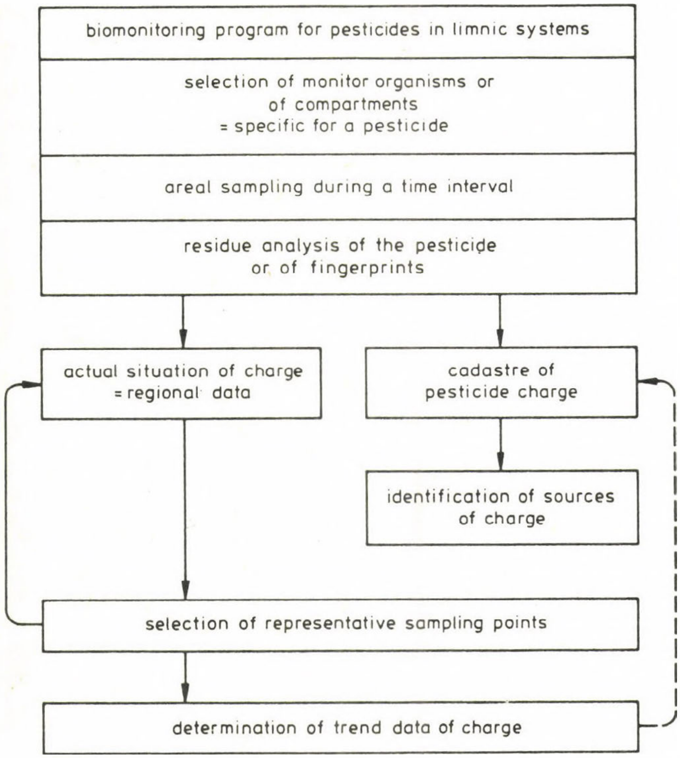


Fig. 1: Schematic diagram of a pesticide monitoring program.

as sediments. This often leads to a restriction of ecosystem use (e.g. for fishing, water use for agriculture, spreading of sediments). The behaviour of heavy metals in limnic systems is characterized by many processes (Fig. 2). The input of metals must be differentiated into two parts - some metals settle in the sediments, whereas other dissolved heavy metals react in water to different binding forms, e.g. chelated heavy metals (with organic or anorganic ligands), adsorbed heavy metals (mainly to clay and to bacteria), heavy metals in organic matrix after bio-uptake, and coprecipitated heavy metals with iron oxides. Dissolved and chelated heavy metals remain in the water body or reach the outflow, whereas the other binding forms are deposited in the sediments. Between the overlying water and sediments, some processes like cation exchange, decomposition of organic material and release of heavy metals, as well as the redox chemical mobilization of coprecipitated heavy metals with iron oxides can occur (Gunkel 1986, Gunkel & Sztraka 1986), which lead to release of heavy metals from the sediment and, thus, internal recycling. These recycling processes are of great interest in shallow lakes. Only if the redox potential is less than 200 mV does relatively stable deposition occur.

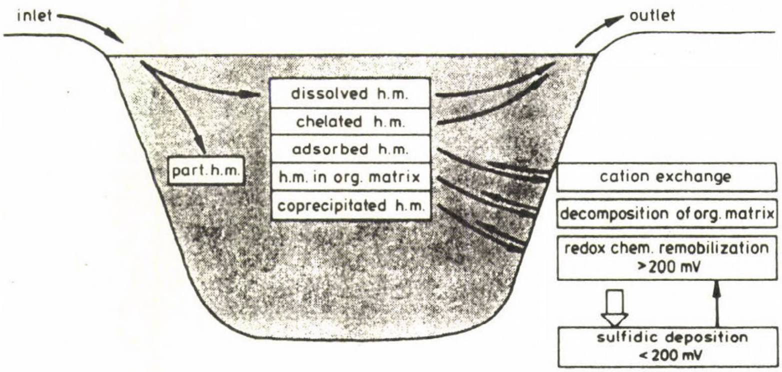


Fig. 2: Diagram of the behaviour of heavy metals in a limnic system.

The charge of heavy metals in a limnic system often is evaluated from different points of view. Heavy metal input can be quantified as a deposition rate ($mg/m^2 \cdot a$), as a concentration, or load (calculated in mg/l or kg/a), as well as an effect (bio-uptake in mg/kg or physiological effect in % of control). The term 'charge of an ecosystem' is still used in this different sense.

2. BIOMONITORING STRATEGY FOR HEAVY METALS IN LIMNIC SYSTEMS

A biomonitoring strategy in a limnic system must be based on the hierarchy of the ecosystems, i.e. the trophic level, and the time dimension as an integral of bio-uptake processes. Depending on the structure of the pesticide, we were able to verify a food chain effect as well as time dependence of bio-uptake. The schematic diagram (Fig. 3) presents organisms with a distinct feeding strategy as well as compartments of a limnic system that are dependent on the ecosystem structure and the time dimension.
Short-term monitoring can be done by water sample evaluation only, but the amount of dissolved heavy metals underlie reactions with high dynamics (i.e. adsorption to microparticles, or uptake by bacteria). Using a new parameter for evaluation, namely the biologically reactive heavy metals (= heavy metals incorporated in organisms or available for organisms; see below), the competition between bio-uptake by bacteria and by other organisms is taken into consideration, and the metal content of the water is not influenced by the presence or absence of bacteria.

Median-term monitoring can be done by evaluating heavy metal residue of organisms, but many organisms undergo complex bio-uptake reactions. Filter feeders like Mytilus or Dreissena, for example, take up metals directly from the water as well as from bacteria and phytoplankton as food particles. They also

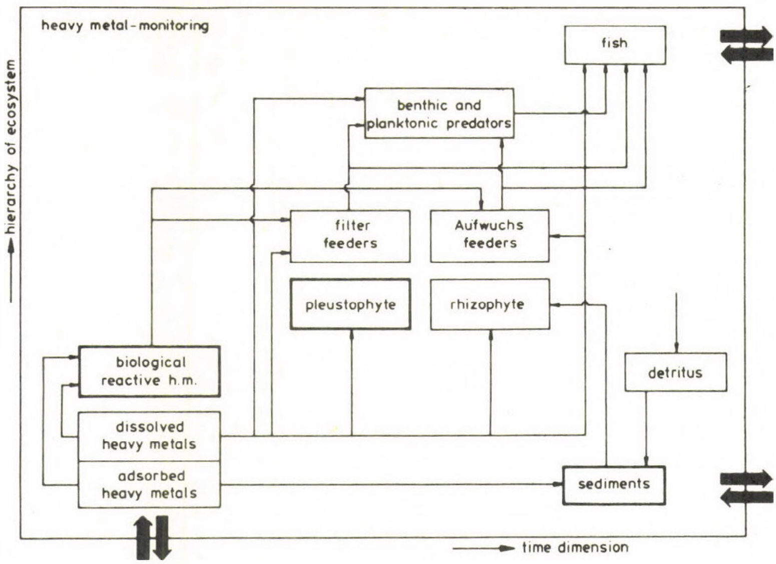


Fig. 3: Schematic diagram of the monitoring strategy for heavy metals in a limnic system and the heavy metal fluxes.

take up metals via seston particles when they are not rejected. Macrophytes of the of pleustophyte type (e.g. Lemna, the duckweed) accumulate metals only from the water, because they are not rooted in the sediment.

Long-term monitoring can be done by sediment evaluation, a well-known method. The residue of heavy metals in sediments is determined in conjunction with the amount of clay (fraction less than 2 µm), consequently the residue data must be calculated for the clay fraction or the clay/silt fraction.

A heavy metal monitoring program must include an evaluation of three compartments - the biologically reactive metals in water, the residue in Lemna, and the heavy metal concentration in sediments. This enables one to register and evaluate the input, the effective concentration in the ecosystem, and the effect of heavy metals with regard to bio-uptake by organisms.

2.1. Heavy metal residue in water

Heavy metal speciation in water can be determined by a method of sequential extraction (Tab. 1). Thus, it is possible to distinguish dissolved heavy metals, adsorbed heavy metals, chelated heavy metals, heavy metals incorporated in bacteria and other small organisms as well as heavy metals in a particulate form (as residual fraction; Schulze & Gunkel 1988). Dissolved, adsorbed, chelated and incorporated heavy metals are defined and evaluated as biologically reactive heavy metals. In Table 2 the heavy metal content (Zn, Cu, Cd) as well as the speciation is given for 18 lakes in Berlin(West). Only small amounts of dissolved heavy metals are detectable, yet a significant amount must be evaluated as biologically reactive (30 to 50 % of the total metal content). A large amount of heavy metals is in a particulated bound form that is not filterable (60 to 80 %). Moreover, we determined a high content of heavy metals in a particulate form (e.g. metal oxides or precipitation particles, 25 to 70 %).

Tab. 1: Method of sequential extraction of heavy metals in water

total heavy metals	H_2O_2/HNO_3 - treatment
dissolved heavy metals	dialysis (13,000 mol) filtration (0.45 µm)
adsorbed heavy metals	0.05 mol/l $CaCl_2$
chelated heavy metals	0.05 mol/l EDTA
incorporated heavy metals	121 °C, 1.3 bar; EDTA
biologically reative heavy metals	sum of fractions
particulate heavy metals	residual fraction

A comparison of dissolved heavy metals and biologically reactive heavy metals is given in Fig. 4. Naturally, the concentration of biologically reactive metals was higher than the concentration of dissolved heavy metals. In many samples, no dissolved heavy metals (e.g. Cd) were detectable, whereas the concentration of biologically reactive metals was significantly higher. The determination of the biologically reactive heavy metals in water thus enables a better evaluation of the heavy metal charge of an ecosystem (determination of the concentration of heavy metals in the water, calculation of magnification factors water/sediment or water/organisms).

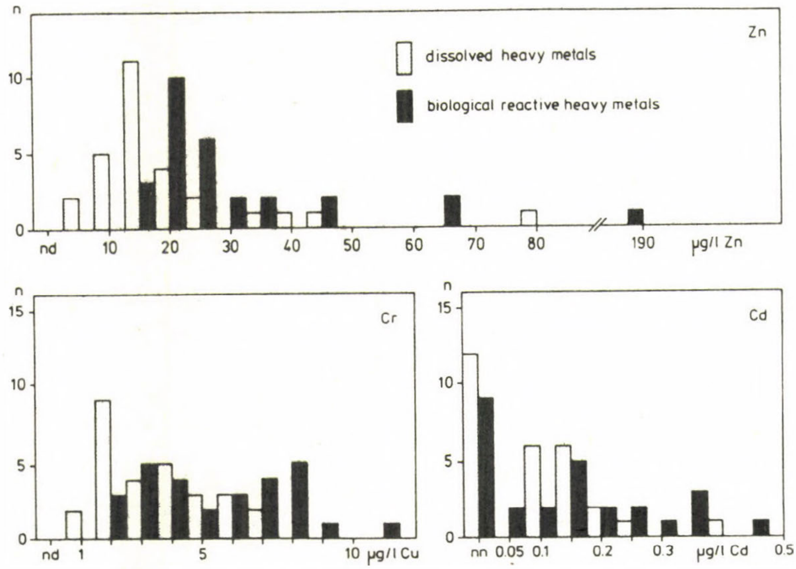
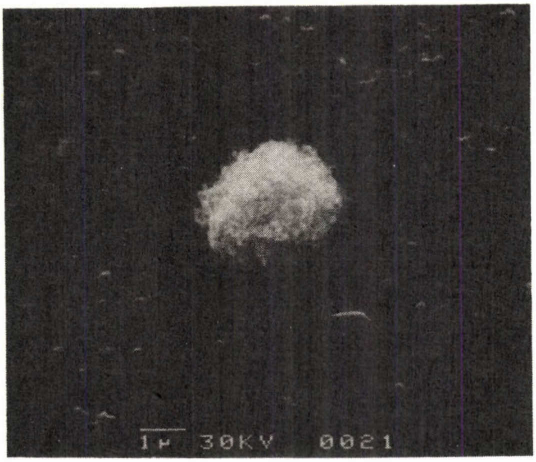


Fig. 4. Comparison of dissolved and biologically reactive heavy metals in waters of Berlin(West).

The other heavy metal binding state of great significance in lakes in Berlin(West) is particulate metals (Tab. 2). We used scanning electron microscopy with energy dispersive X-ray microanalysis and proved that iron oxidizing bacteria of the genus Siderocapsa treubii contained, in addition to iron (and sometimes manganese), heavy metals (Gunkel 1986). The capsules



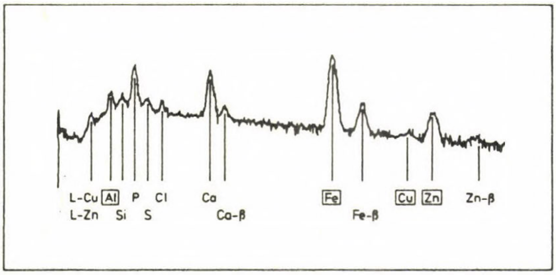


Fig. 5: Scanning electron microscopic view of a cell of <u>Siderocapsa treubii</u> (above) and energy dispersive X-ray microanalysis (below).

of the cells of <u>Siderocapsa</u> contained Si, P, S, Cl, Ca and Fe. Besides, we found aluminum as well as zinc in high concentrations, and copper in small concentrations (Fig. 5). The maximum sensitivity of this method is in the range of iron, being only a half-quantitative analysis. Up to now, we have found many cells and colonies of iron oxidizing bacteria containing heavy metals. We also found particles resembling like detritus that contained metals, which we must identify as a coprecipitation of heavy metals with iron oxides or with other precipitation products. The crusts are deposited in the sediment and can be redissolved via redox chemical reaction, and iron as well as heavy metals are introduced to the water body (Fig. 2).

Tab. 2: Heavy metal concentrations and speciations in waters of Berlin(West), in µg/l, n.d. = not detectable.

		total h. m.	dissolved h. m.	adsorbed h. m.	particulate h. m.	biol. reactive h. m.
Zn	m	41.5	17.5	24.6	10.3	22.3
	s	36.0	15.3	24.5	13.3	8.0
	min	11	4	n.d.	n.d.	11
	max	198	81	117	43	186
Cu	m	12.7	3.2	9.6	7.3	5.4
	s	10.5	1.8	10.4	9.4	2.4
	min	2	1	n.d.	n.d.	2
	max	48	7	47	37	11
Cd	m	0.42	0.07	0.35	0.28	0.13
	s	0.29	0.09	0.28	0.27	0.13
	min	n.d.	n.d.	n.d.	n.d.	n.d.
	max	1.1	0.4	0.9	0.8	0.4

2.2. Heavy metal residue in macrophytes

Median-term monitoring can be done by evaluating macrophytes, especially plants of the pleustophyte type, because they accumulate heavy metals only directly from the water (Fig. 6). Rhizophytes are rooted in the sediment and accumulate heavy metals from the water, sediment, and interstitial water; besides an excretion of heavy metals can occur via loss of leaves and via ion excretion of plants. Representative pleustophytes in the area of Berlin(West) are Lemna minor and L. gibba, which are wide-spread. The determination of the bioaccumulation factors for heavy metals in the lakes of Berlin(West) lead to a factor of about 10,000 (Zn, Cu, Cd), but the use of bioaccumulation factors is not correct: The heavy metal concentration in water is highly dynamic (adsorption and

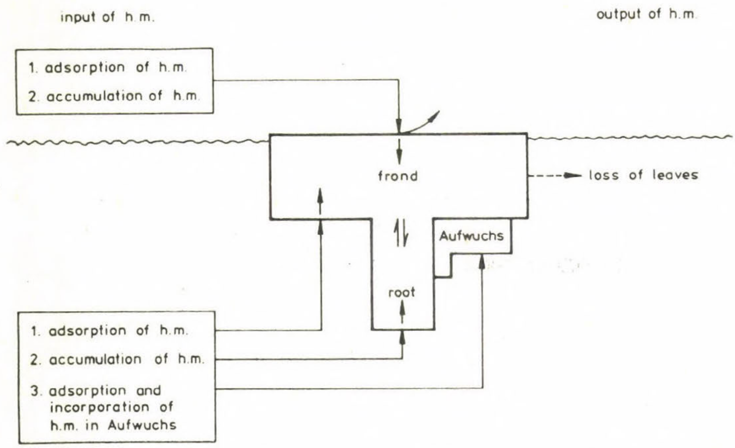


Fig. 6: Schematic diagram of heavy metal fluxes in pleustophytes.

uptake by bacteria), whereas the residue in pleustophytes is less changeable and, besides, the bioaccumulation factor is only applicable under steady state conditions, which are not realizable in field systems. It is better to evaluate the absolute residue in the macrophytes, and by using the four-sigma-rule, the median concentration of the residue can be calculated. This median represents the actual local situation. This procedure enables one to define limited values as well as exceeded values (Fig. 7). The median residue of zinc amounted to about 110 mg/kg, and only a few exceeded values up to 1,000 mg/kg were found. For copper, a median residue of 10 mg/kg with a small scattering of the data was determined, and some exceeded values up to 55 mg/kg were observed. The median cadmium residue was about 0.2 mg/kg with high significance, but many exceeded values were observed extreme residue data ranged up to 4 mg/kg.

Using this method of determination of residue data for pleustophytes, it is possible to quantify the heavy metal charge over a period of some weeks (depending on the life period) and to determine trend data for heavy metal charge.

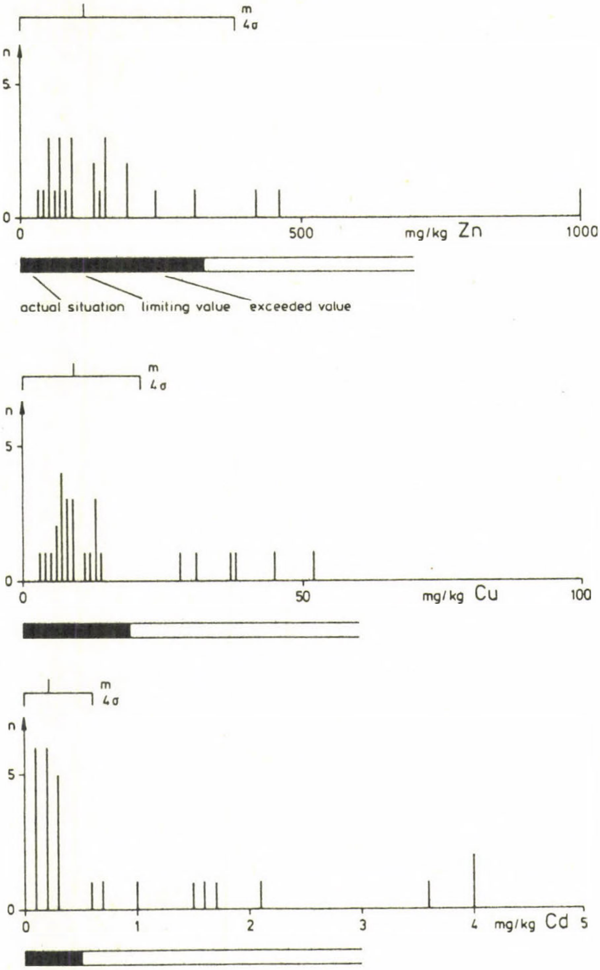


Fig. 7: Heavy metal residue in <u>Lemna minor</u> in waters of Berlin(West).

2.3. Heavy metal residue in organisms

In some organisms, significantly increased residue of heavy metals are found, but heavy metal fluxes in organisms often are complex (way of input and uptake efficiency, site of deposition in organisms, excretion mechanisms). Many investigators use filter feeders, mainly Mytilus or Dreissena, as indicator organisms (Schulz-Baldes 1982, Borchardt 1983), but the metabolic activity of an individual organism as well as pseudo-faeces excretion lead to data being less representative.

Residues in fish, e.g. in the dorsal muscle, are also used as an indicator of the heavy metal charge. Mainly for cadmium (besides mercury) and lead, background values can be determined, and exceeded values as well as trend data are available. These data indicate an increase in the charge of cadmium in limnic systems in WestGermany with a median residue of 0.07 mg/kg (s = 0.08) in fish and exceeded values of up to 0.86 mg/kg (sources: ZEBS 1984).

2.4. Heavy metal residue in sediments

Determination of heavy metal residue in sediments or in the clay fraction of sediments is a well-known method for monitoring and surveying ecosystems. High concentrations of heavy metals are usually found in sediments. In lake sediments of Berlin(West), zinc concentrations of 80 to 2,000 mg/kg dry matter were observed; the copper concentrations were 25 to 2,500 mg/kg, and the cadmium concentrations ranged from 0.3 to 25 mg/kg (Umweltatlas Berlin 1986).

The significance of the heavy metal residue in limnic sediments is limited by some ecological processes of high importance in shallow and eutrophic waters (Fig. 8). The heavy metal concentration in sediments is determined by

a) exchange processes between sediment and water body, mainly iron exchange, precipitation, and mobilization;

b) sediment removal by flood water and gravity currents; and

c) sedimentation rate as a product of autochthonous production and decomposition and of erosion in the catchment area. Particularly the eutrophication processes in limnic systems lead to an increased sedimentation rate with the corresponding

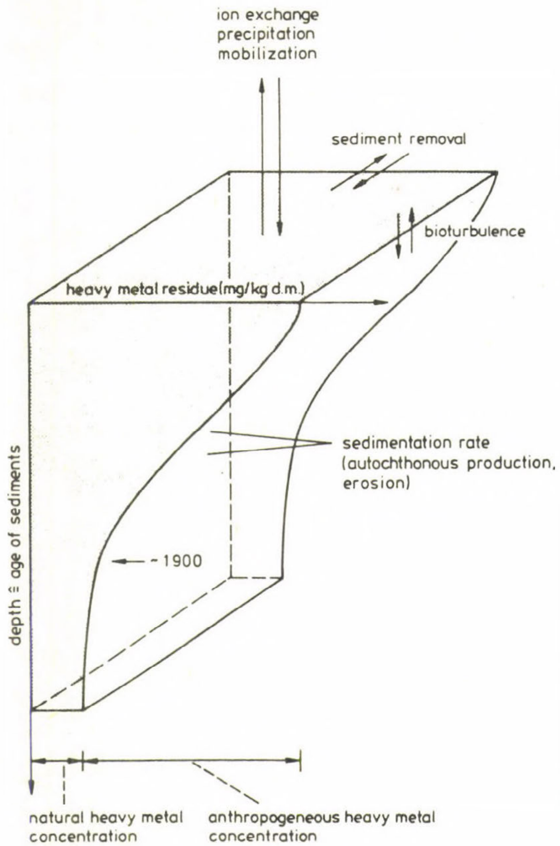


Fig. 8: Schematic diagram of heavy metal fluxes in sediments of limnic systems and processes influencing the metal concentration.

dilution of heavy metals in sediments. Consequently, the stratification of sediments with regard to the age of the sediments is limited by these processes and leads to a dating of several years, and trend data are primely available after some decades.

3. CONCLUDING REMARKS

Biomonitoring of pesticides in limnic systems is of high significance, because many ecosystems are contaminated - mainly in industrial areas and highly populated areas. It is necessary to quantify the pesticide charge as well as to determine trend data, e.g. as feedback for water management.

In routine survey programs done up to now, the total content of heavy metals in water and of the dissolved heavy metals were registered as well as the content of heavy metals in sediment. It was, however, impossible to distinguish the input of metals, the deposition rate of particulate metals, and the mobilization of metals in the limnic system. The amount of heavy metals dissolved in the water must be regarded as a competition reaction with regard to uptake by bacteria and accumulation in organisms. This competition reaction should be considered when determining the parameter for biologically reactive heavy metals and for bio-uptake by organisms. The heavy metal content of sediment must supplement these data.

References

Borchardt, T.: Influence of food quantity on the kinetics of cadmium uptake and loss via food and seawater in Mytilus edulis. - Mar. Biol. 76, 67-76, 1983.

Förstner, U. & G.T.W. Wittmann: Metal pollution in the aquatic environment. 2nd ed., pp. 473, Springer Verlag 1981.

Gunkel, G: Untersuchungen zum Verhalten von Schwermetallen in Gewässern. I. Die Bedeutung eisenoxidierender Bakterien für die Kopräzipitation von Schwermetallen. - Arch. Hydrobiol 105, 489-515,.1986.

Gunkel, G & A. Sztraka: Untersuchungen zum Verhalten von Schwermetallen in Gewässern. II. Die Bedeutung der Eisen- und Mangan-Remobilisierung für die hypolimnische Anreicherung von Schwermetallen. - Arch. Hydrobiol. 106, 91-117, 1986.

Gunkel, G.: Mechanismen der Aufnahme und Verteilung von organischen Schadstoffen in aquatischen Organismen. pp. 39-68. In: Lillelund, K., et al. (eds.): Bioakkumulation in Nahrungsketten. pp. 327, VCH, 1987.

Mitra, S.: Mercury in the ecosystem. Its dispersion and pollution today. pp. 327. TransTechPublications Switzerland, 1986.
Schulz-Baldes, M.: Muscheln als Monitororganismen für die Schwermetallbelastung im Meer. pp. 83-92. In: Ernst, W. (ed.): Meeresverschmutzung. - Campus Verlag Frankfurt, 1982.
Schulze, G. & G. Gunkel: Verteilung und Umsetzungen von Schwermetallen in der biologischen Stufe einer kommunalen Kläranlage. - Vom Wasser 70, 209-220, 1988.
Umweltatlas Berlin: Senator für Stadtentwicklung und Umweltschutz, Berlin. Bd. 1, 01 Boden, 1985.
ZEBS: Zentrale Erfassungsund Bewertungsstelle für Umweltchemikalien des Bundesgesundheitsamtes: Arsen, Blei, Cadmium in und auf Lebensmitteln. Hersg.: Weigert, P. u.a., Heft 1/1984.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

BIOINDICATORS IN MONITORING HEAVY METAL POLLUTION IN LAKE BALATON (HUNGARY) AND ITS CATCHMENT AREA

J. SALÁNKI

Balaton Limnological Research Institute of
the Hungarian Academy of Sciences
H-8237 Tihany, Hungary

As is well known, heavy metals are natural, geologically determined elements, which are present everywhere in lower or higher concentrations. Although the amount of metals is constant on the earth, their biologically available concentration is permanently increasing in the environment as a result of geological weathering and human activity. Some of the metals are also incorporated in living organisms as essential components while others are considered as toxic abiotic elements. Toxic heavy metals represent an important constituent among environmental pollutants that can have unfavourable influences on living organisms (Salánki, Salama 1987).

In a lake ecosystem the danger of heavy metals is strengthened by two circumstances: (a) from a polluted catchment area heavy metals proceed permanently into the lake through waterflow, and, as a result, the concentration may increase progressively in the lake water and bottom; (b) aquatic organisms accumulate heavy metals, which may be dangerous for both accumulator species and consumer organisms. Thus, heavy metal pollution is an important issue in lake protection and management.

In this respect we are studying Lake Balaton and its environment. This is a large shallow lake in Hungary (Fig. 1). Its surface is nearly 600 km^2 with a catchment area of 6000 km^2. The catchment area is rather peculiar: besides 21 small streams that run into the lake, there exists but a single river, the Zala, which is the lake's principal water supply; there is an outlet, the Sió Canal, which regulates the water level. The vicinity of Lake Balaton is Hungary's main recreational area.

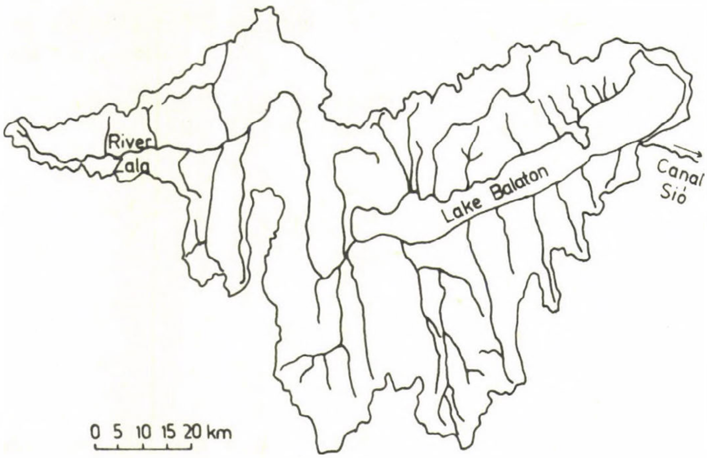


Fig. 1. Lake Balaton and its catchment area.

It is used, moreover, for fishing and as a source of drinking and irrigation water. The lake's protection is a great concern to the government and a cause of public anxiety.

When discussing the heavy metal pollution in the area of Lake Balaton one should take into consideration the possible sources of metals in the region. For Lake Balaton, as in other cases, some of these sources have more, others less significance.

The background heavy metal concentrations in this region are the results of normal geological weathering.

Ores and metals are neither mined nor processed in the Balaton area. The only metal-containing ore in the vicinity is bauxite; however, it lies outside of the catchment area.

The use of metals and metal compounds in industry and agriculture is a more serious question. Although no significant industry exists in the region except for a chemical factory and a paper mill, there are a number of small industries of local interest which release heavy metals into the environment. These function as point sources of heavy metal pollution. Furthermore the use of heavy metals in agricultural technology is not controlled properly - they are partly components of chemicals used against fungi and pests, partly fertilizers containing heavy metals as a contamination. Since agriculture is intensive in the Balaton area - vineyards on the north, crop and wheat fields on the south shore - the effect of agriculture is remarkable.

The leaching of metals from garbage and solid waste dumps should also be considered as a main source of metal pollution. Waste deposits in the Balaton area are rather numerous and also rather uncontrolled.

Domestic effluents and urban sewage water are also important sources of heavy metals in the region. In the 3400 km^2 area of the 200 km long lakeshore there are 150 settlements with 250,000 permanent inhabitants. This number nearly triples during the tourist season. Recently measures have been taken to direct waste water away from the catchment area, but several years ago a large number of sewage plants were located close to the lake; some of them are still functioning.

Atmospheric sources of metal pollution are partly at an uncontrollable distance; however, they are also present in the region as emissions of dust and ashes. Road and rail traffic, which increase greatly during the tourist season, should also be mentioned in this category.

In order to obtain a true picture of heavy metal pollution in the region, we started nearly 10 years ago to measure the concentration of heavy metals in different aquatic animals living both in the lake and the catchment area.

It is known that there are differences among organisms in susceptibility to heavy metals and also in their capacity to accumulate them. Therefore, when using animals, we wanted not only to check the level of pollution, but also to select good biomonitors for further use. The results of these investigations, which were conducted together with my colleagues, have been partly published.

MATERIALS AND METHODS

Among the organisms used for the investigations were planktonic animals (Crustacea), benthic animals (mussels, Chironomidae larvae), aquatic snails, and predatory and nonpredatory fish. In the case of mussels and fish not the total animal but separate organs were used for heavy metal measurements.

Concentrations of Hg^{2+}, Cd^{2+}, Ni^{2+}, Pb^{2+}, Cu^{2+}, Zn^{2+} were measured by atomic absorption spectrophotometry (Salánki et al., 1982). Their values are given in the tables and figures to dry weight.

RESULTS AND DISCUSSION

Studies of the heavy metal concentrations in Crustacea plankton in different open water locations showed an uneven distribution. In general, the values were higher in the western basin and lower in the eastern part. The reason may be that the Zala river falls into the lake in the west while the Sió outlet is situated in the east. Nevertheless, there is a difference between the peaks of various metals (Fig. 2), even at different locations of the same region, indicating a clear distinction concerning their origin. In contrast to a rather uniform distribution of mercury in the westernmost part, cadmium is highest near the town of Keszthely. Lead is highest in the second section, in which there are streams from the northern area that enter the lake and the outlet of a sewage works. The distribution of Cu and Zn concentrations in the Crustacea plankton was similar to Cd distribution (V.-Balogh, Salánki 1984).

A survey of the heavy metal concentrations in various animals and organs showed very precisely that there are differences between the bioaccumulation of metals in various organisms and tissues. It became clear that mercury is concentrated better in Chironomidae larvae than in Crustacea plankton. Furthermore, among organs the gills of mussels and the kidneys of fish proved to be prominent accumulators of mercury, while accumulation of zinc was most extensive in the gills of mussels (Fig. 3). Differences in the concentration of Cd, Cu and Pb were also found between species and organs (Salánki et al. 1982). In general, of the investigated animals mussels proved to be the best accumulators for each of the heavy metals, while among organs the gills of mussels, and the gills and kidneys of

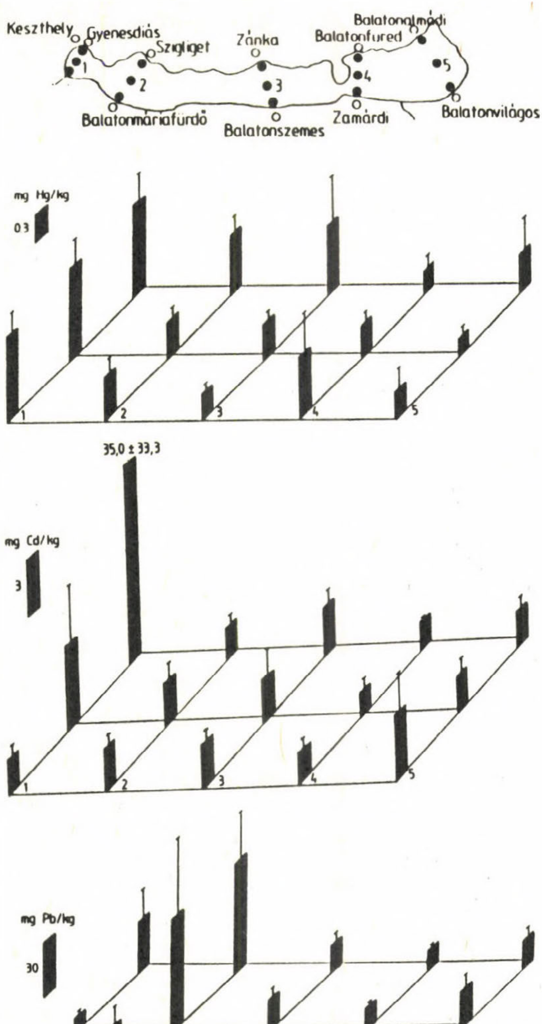


Fig. 2. Mercury, cadmium and lead concentrations in the Crustacea plankton at different parts of Lake Balaton. Locations where samples were collected are marked at the map by full circles. Sections are numbered from 1 to 5.

fish species are the best biomonitors for heavy metal pollution.

Special investigations were carried out in animals in the Zala river, the main water source of Lake Balaton. Two places

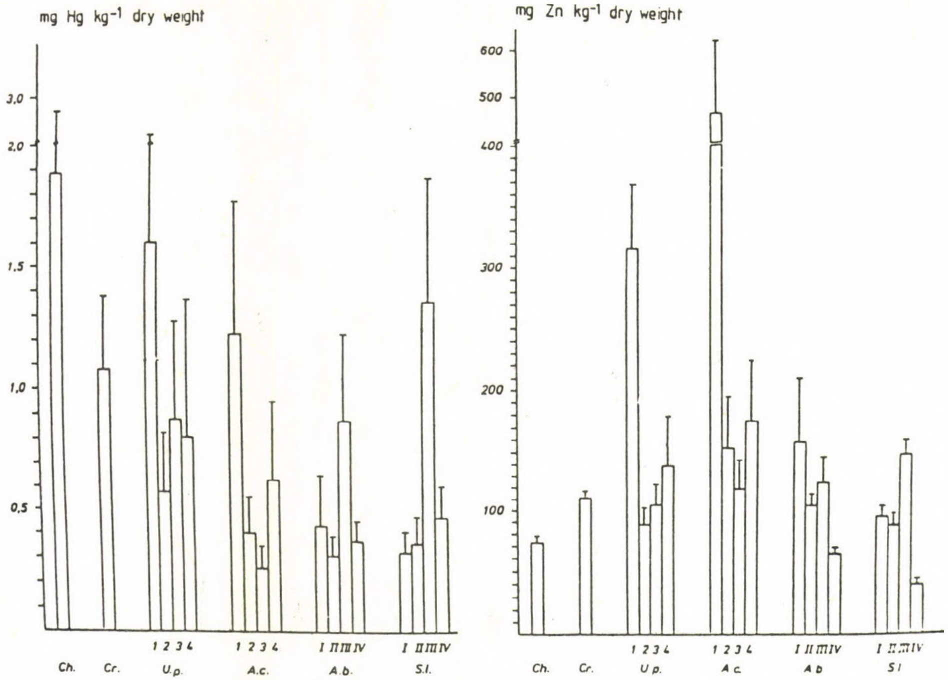
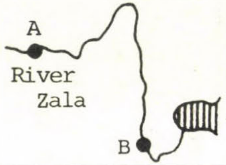


Fig. 3. Concentrations of mercury and zinc in different amimals collected from the open water of Lake Balaton. Ch - Chironomidae larvae; Cr - Crustacea plankton; U.p. - Unio pictorum; A.c. - Anodonta cygnea; A.b. - Abramis brama; S.l. - Stizostedion lucioperca; 1 - gill, 2 - foot, 3 - adductor muscle, 4 - mantle; I - gill, II - liver, III - kidney, IV - muscle.

were selected, A - above a large town (Zalaegerszeg) and B - below the town. Heavy metals were measured in Gammarus (Crustacea) during spring and summer. Parallel samples were taken from the water and from the animals. It became obvious that, although in a number of cases the metal concentration is higher closer to the lake, there are opposite instances as well (Table 1). For cadmium, lead and zinc the change of concentration in the water was compared in the animals. However, the change of copper, mercury and nickel concentrations between sites A to B was not reflected in the metal concentration of Gammarus.

Table 1. Heavy metal concentrations in the water and the Crustacea plankton (Gammarus) in the Zala river above (A) and below (B) the town Zalaegerszeg, during spring and summer. Mean of 3-5 measurements



			Cu	Cd	Hg	Ni	Pb	Zn
spring	Water μg/l	A	2.92	0.04	0.3	3.50	0.73	<5
		B	3.22	<0.01	<0.1	2.4	0.82	13
	<u>Gammarus</u> <u>roeselii</u> G. mg/kg	A	74.4±9.46	22.1±2.91	0.329±0.015	12.8±1.23	11.8±1.53	55.9±7.65
		B	79.6±3.47	12.0±3.59	1.15±0.065	14.2±2.64	27.9±8.20	82.3±9.04
summer	Water μg/l	A	3.09	<0.01	<0.1	3.4	<0.1	11
		B	2.68	<0.01	<0.1	1.7	0.36	9
	<u>Gammarus</u> <u>roeselii</u> G. mg/kg	A	55.7±7.30	22.1±0.100	<0.070	36.0±3.35	<5	85.1±4.93
		B	150±69.4	59.9±23.9	2.42±1.61	42.8±16.0	98.8±31.3	76.0±25.0

Note: values are given for the water in μg/l, for animals in mg/kg.

A similar observation was made when metals were measured in the snail Planorbarius and in the gill of Anodonta, which were collected at the mouth of the Zala river. In most cases the metal concentrations in the water were different in summer and in spring. Copper and zinc concentrations closely followed these changes in the animals (Table 2). A possible explanation for these divergences is that in the water sample one determines the actual concentration at the time of sampling, while in animals an integrated metal concentration - which reflects the level of pollution of a longer period - is measured. This latter is an advantage of using biomonitors instead of water samples only. As a rule water sampling cannot disclose an episodic pollution wave if it occurs between two samplings.

In the course of our measurements we could determine also the rate of bioaccumulation of various metals in different animals and organs. Bioaccumulation can be expressed as a concentration factor, which indicates the ratio of the metal concentration in the animal and in the water (Taylor 1983). Table 3 summarizes some of the concentration factors we found in field measurements when water pollution was comparatively low. It is questionable whether these values will change when the pollution in the water increases. In any case all the tested animals seem to be good accumulators for the investigated metals.

Great differences in the heavy metal content of the snail Lymnaea stagnalis, which was obtained from various ponds and reservoirs around Lake Balaton, were found, indicating differences in the pollution of their habitat (V.-Balogh et al. 1988). We are also extending our research to fish species living in the Zala river and in various streams in the catchment area (V.-Balogh et al., this Volume) with the objective of disclosing the heavy metal pollution of given regions. Two fish species were investigaged, Alburnus alburnus and Carassius auratus, depending on their availability in different streams. Clear differences were observed at different locations for each of the metals, referring to differences in the heavy metal loading of various habitats. Nevertheless, with fish, migration must be also taken into consideration, as it can be a source of error in evaluating local pollution.

Table 2. Heavy metal concentrations in the water, in the snail Planorbarius and in the gill of the mussel Anodonta collected at the mouth of the Zala river

Mouth of Zala river		Cu	Cd	Hg	Ni	Pb	Zn
Water	spring	1.35	<0.01	0.15	2.9	<0.66	7
μg/l	summer	1.59	<0.01	0.15	1.7	<0.1	13
Planorbarius corneus L.	spring	12.1±2.01	4.91±0.489	0.106	9.79±1.06	<5	50.3±3.76
mg/kg	summer	43.7±21.6	21.5±6.00	0.889±0.131	10.9±2.54	8.68	102±50.9
Gill of Anodonta anatina L.	spring	22.5±5.05	8.35±1.85	0.508±0.145	26.3±9.20	51.9±13.6	481±205
mg/kg	summer	50.3±19.9	42.8±9.36	0.840±0.170	17.9±2.01	31.6±2.25	461±37.5

Table 3. Concentration factors for four invertebrate species in the Lake Balaton area

Species	Cu	Cd	Hg	Ni	Pb	Zn
Gammarus roeselii G.	$2.5\cdot10^4$	$1\text{-}5\cdot10^5$	$10^3\text{-}10^4$	$3\text{-}6\cdot10^3$	$1\text{-}3\cdot10^4$	$6\text{-}9\cdot10^3$
Planorbarius corneus L.	$9\cdot10^3\text{-}3\cdot10^4$	$5\cdot10^5\text{-}2\cdot10^6$	$7\cdot10^2\text{-}6\cdot10^3$	$3\text{-}6\cdot10^3$	$8\cdot10^4$	$7\cdot10^3$
Viviparus contectus M.	$4\cdot10^4$	10^5	$3\cdot10^3$	$3\cdot10^3$	$10^3\text{-}10^5$	$2.5\cdot10^4$
Anodonta anatina L. (gill)	$1\text{-}3\cdot10^4$	$4\text{-}8\cdot10^5$	$3\text{-}5\cdot10^3$	10^3	$3\text{-}5\cdot10^4$	$3\text{-}7\cdot10^4$

CONCLUSIONS

The relative degree of heavy metal pollution can be determined with great accuracy by using biological test organisms. The differences in heavy metal concentrations in the same animal species or organ collected in different places refer to different levels of pollution. Through the analysis of biomonitor species, the sources of heavy metal pollution can be discovered.

Heavy metal pollution in Lake Balaton is heterogeneous. The most polluted part is the western basin where the Zala river enters the lake and the town of Keszthely is situated.

Using bioindicator organisms a permanent control can be established in each region where protection from heavy metal pollution is of great importance, and this system can be part of a conservation and management programme in lake ecosystems. For economic reasons, only the best accumulators should be regularly controlled, and in aquatic ecosystems gills of mussels are recommended for this purpose.

REFERENCES

Salánki, J., Salama, H.S. (1987): Signalization, monitoring and evaluation of environmental pollution using biological indicators. Acta Biol. Acad. Sci. Hung. 38, 5-11.

Salánki, J., V.-Balogh, K., Berta, E. (1982): Heavy metals in animals of Lake Balaton. Wat. Res. 16, 1147-1152.

Taylor, D. (1983): The significance of the accumulation of cadmium by aquatic organisms. Ecotox. Envir. Safety 7, 33-42.

V.-Balogh, K., Salabarria Fernandez, D., Salánki, J. (1988): Heavy metal concentrations of *Lymnaea stagnalis* in the environs of Lake Balaton (Hungary). Wat. Res. 22, 1205-1210.

V.-Balogh, K., Salánki, J. (1984): Use of crustacea plankton in evaluating heavy metal pollution of Lake Balaton. J. Hydrobiology (USSR) 20, 56-64 (in Russian).

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

HEAVY METALS IN ORGANISMS OF THE ESTUARINE AREA SURROUNDING THE MOUTH OF THE SARAMAGUACÁN RIVER, CUBA

D.M. SALABARRIA FERNANDEZ, K.V.-BALOGH,* J. SALÁNKI*

National Commission for Environmental Protection,
Cuban Academy of Sciences, Habana, Cuba
*Balaton Limnological Research Institute of the Hungarian
Academy of Sciences, H-8237 Tihany, Hungary

ABSTRACT

Concentrations of Cr, Cd, Pb, Cu, Fe, Mn, Zn and Ni were measured in oysters, two species of estuarine gastropods, two species of fishes, in seaweeds, two species of seagrass, and in terrestrial plant Rhizophora mangle, as well as in sediments, using atomic absorption spectrophotometry.

Highest concentrations for Cd, Pb, Cu, Zn and Ni, were found in molluscs, both gastropods and oysters. Most Cr, Fe and Ni occurred in sediments, seaweeds and seagrass species, while highest values for Mn were found in seagrass species. For all metals, lowest values occurred in fishes.

The results were compared with those obtained for similar species living in other estuaries and it is concluded that our research area belongs to moderately polluted regions.

INTRODUCTION

Heavy metals have been identified as a major element of contamination, affecting not only the natural environment, but also the fishing industry and human health.

The toxicity and bioaccumulation of these metals in aquatic organisms have been studied extensively in the last decades and some authors have demonstrated the positive use of some organisms as biological indicators.

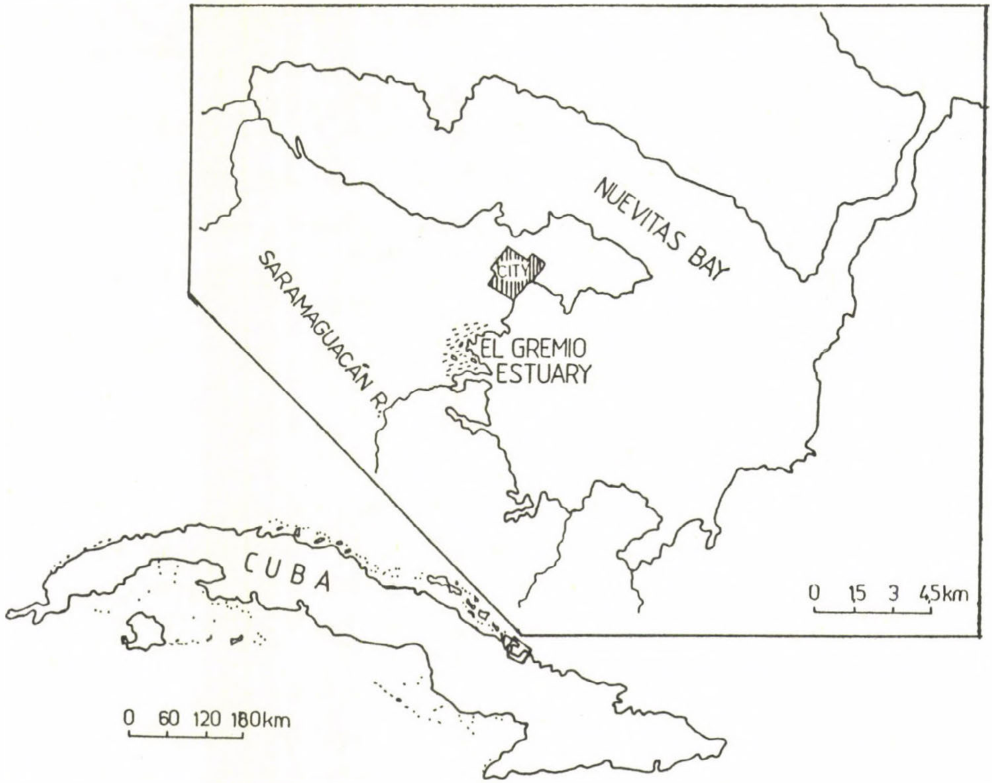


Fig. 1. Nuevitas Bay, El Gremio Estuary location

The aim of our work was to evaluate heavy metal pollution in an estuarine ecosystem from Nuevitas Bay, using different species (plants and animals), as well as the sediment.

Nuevitas Bay is located in the northeast part of Cuba (Fig. 1). It is an important area for industrial development, harbour activities and fishing. El Gremio Estuary, in the inner part of the bay, is a marshland area, very shallow, located around the Saramaguacán River. Near to this place wastes are discharged from a wire and electrode factory releasing various heavy metals, and also municipal wastes, which partially reach the coast by run-off.

MATERIALS AND METHODS

The oyster *Crassostrea rhizophorae*, gastropods *Littorina angulifera* and *Neritina virginea*, fishes *Mugil liza* and *Gerres cinereus*, red seaweeds, seagrasses *Thalassia testudinum* and *Syringodium filiforme*, and leaves and roots of *Rhizophora mangle* were analysed.

Molluscs were collected by hand from mangrove roots, they were scrubbed and shucked, removing the soft parts. Fishes were collected using a fishing trawl net. Macroalgae and seagrass were taken manually from the bottom, and carefully washed to remove sediment and epiphytic species. Surface sediment was obtained by means of a Van Veen grab sampler.

All samples were dried at 105^{o}C during 48 hours, and digested in the presence of concentrated HNO_3 and H_2O_2 for 5 to 6 hours in a water bath.

Samples were adjusted with tridistilled water to 50 ml, and kept in polyethylene flasks until analysed.

Cr, Cd, Pb, Cu, Fe, Mn, Zn and Ni were measured using atomic absorption spectrophotometry. For animal species five replicates were made and three for plants and sediments. All concentration values are expressed in mg/kg dry weight.

RESULTS AND DISCUSSION

According to the literature, heavy metal concentrations are extremely variable in various marine and freshwater organisms (1) depending on the geochemical background and the level of pollution of the given area. The concentrations we found were also very different in various organisms, even in closely related species (Fig. 2).

The highest chromium concentration occurred in sediment, while similar values were obtained for red macroalgae. Relatively high concentrations were also found in seagrass species and in the gastropod *L. angulifera*. Values in other species were low. Some authors (2) stated that the role of Cr in biological systems had been less well established, however, it should not be excluded. They suggested that in some cases the

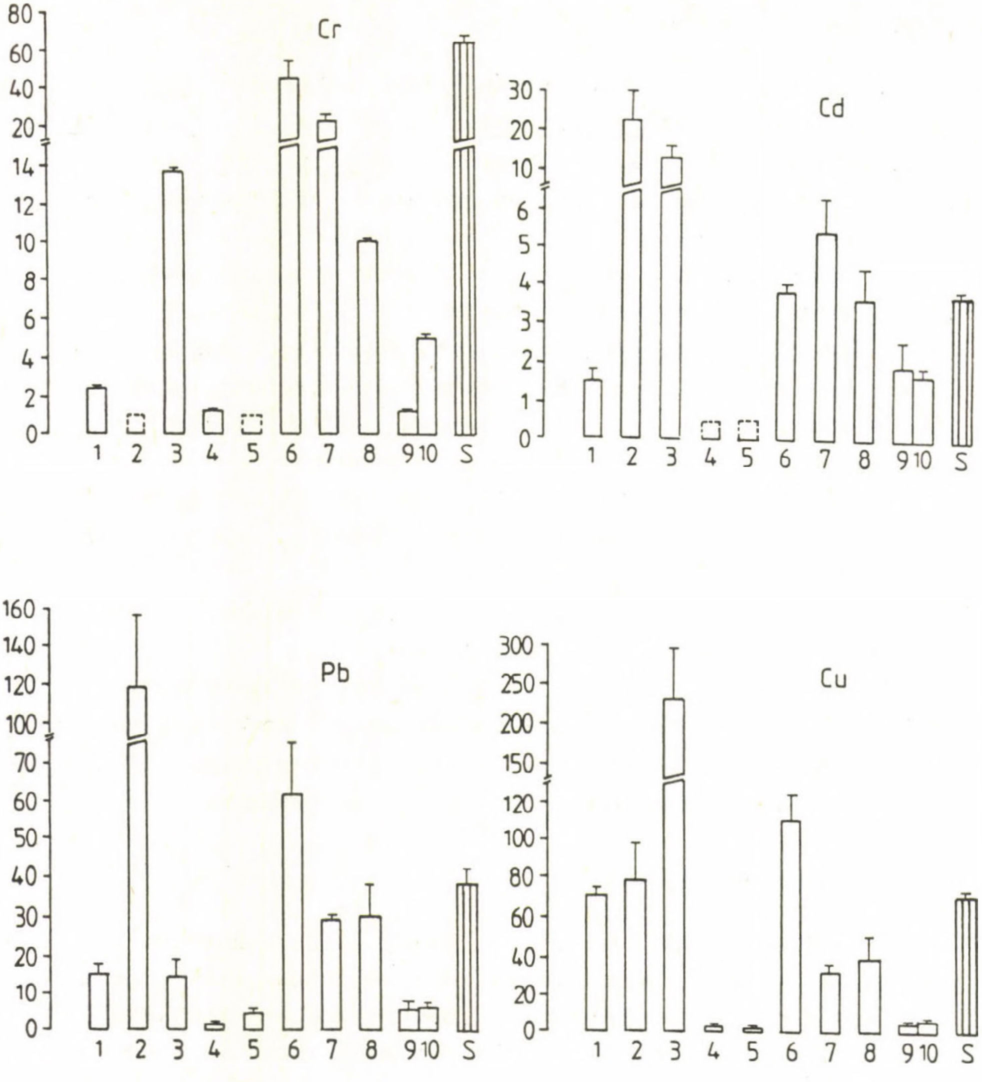
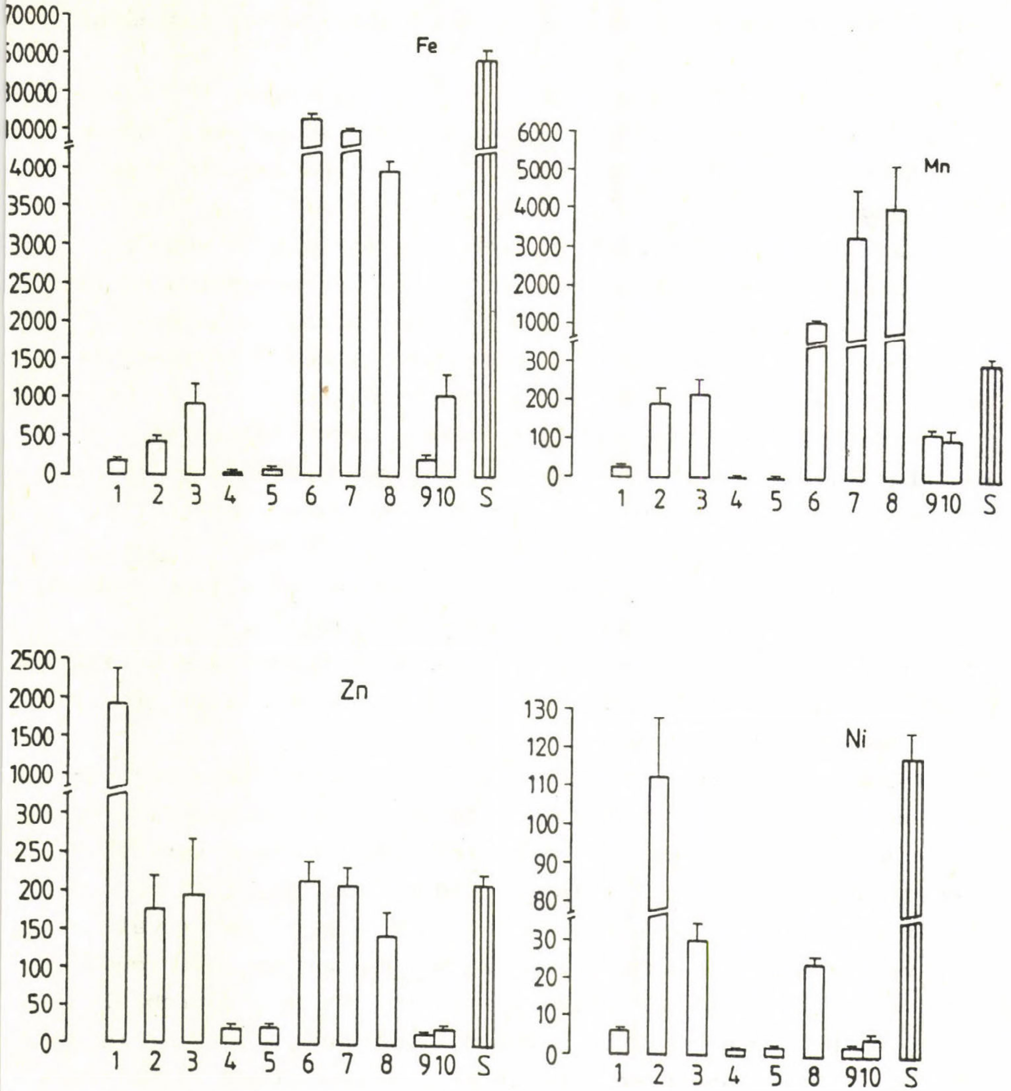


Fig. 2. Heavy metal concentrations in organisms from El Gremio Estuary. 1. Crassostrea rhizophorae; 2. Neritina virginea; 3. Littorina angulifera; 4. Mugil liza;



5. <u>Gerres</u> <u>cinereus</u>; 6. Rhodophyceae; 7. <u>Thalassia</u> <u>testudinum</u>; 8. <u>Syringodium</u> <u>filiforme</u>; 9. Leaves of <u>Rhizophora</u> <u>mangle</u>; 10. Roots of <u>R</u>. <u>mangle</u>. S = sediments

level of chromium in molluscs is due to sedimentary contamination.

Gastropods N. virginea and L. angulifera showed the highest cadmium concentrations, much higher than in sediments, and very different from values obtained in other species. In both fish species, values were below detection limit.

Despite the wide variation, lead concentrations were found to be high for almost all organisms, with the exception of fish species and R. mangle. The highest values occurred in the gastropod N. virginea and seaweeds, much higher than those in sediments, as it occurred for Cd concentrations.

In the oyster C. rhizophorae, levels were higher than those reported in the literature for Crassostrea species (3, 4) and similar to values obtained in Cuba for the same species (5).

Very high copper concentrations were found for L. angulifera, and very different from those reported for affine species (6). High values were also found in N. virginea, C. rhizophorae and seaweeds. These were higher than those in sediments. Relatively high values occurred in seagrass species, while in fish species and R. mangle levels were low.

A great variation was detected in respect of iron concentrations. Extremely high values were found in sediments. Very high concentrations occurred also in seaweeds and seagrass T. testudinum, while S. filiforme showed values less than one order of magnitude as compared with T. testudinum. Relatively high values occurred in roots of R. mangle and in gastropod species. These high concentrations in sediments are probably due to both high natural background level and industrial input.

The highest manganese concentrations were found in seagrass and seaweeds, higher than those in sediments. Gastropod species showed similar values as leaves and roots of R. mangle, and the lowest concentrations were measured in fishes.

The highest zinc concentrations were found in C. rhizophorae according to values reported in the literature for Crassostrea species at places with different degree of pollution (4, 7). Values in seagrass and seaweeds were similar to those in sediments.

Very high nickel concentrations, similar to those in sediments, were found in N. virginea. Relatively high values occurred in the other gastropod L. angulifera and in seagrass S. filiforme while in the other species values were low.

As we can see, concentrations for Cd, Pb, Cu, Mn, Zn and Ni were higher or similar in molluscs, seaweeds and seagrass species than those in sediments, demonstrating metal accumulation in the organisms.

CONCLUSIONS

1. Concerning heavy metal pollution, El Gremio Estuary must be considered as a moderately contaminated area.

2. The highest accumulation values were found for Cd, Pb and Cu in gastropods; for Pb and Cu, seaweeds also showed high values; for Cr, Mn, and Fe, in seaweeds and seagrass; for Zn in oysters; and for Ni in the gastropod N. virginea, as compared to the sediment.

3. Heavy metal concentrations in seaweeds and seagrass reflect the level in the sediments.

4. It is evident that sediments from the estuarine area are enriched mainly with Cr, Pb, Cu, Fe, Zn, and Ni, but there is no obvious impact on the bay; it appears to be restricted to the inner part of the estuary.

REFERENCES

1. Förstner, U., Prosi, F. (1979): Heavy metal pollution in freshwater ecosystems. In: Ravera, O. (ed.): Biological Aspects of Freshwater Pollution. Pergamon Press, Oxford - New York, pp. 129-161.
2. Pringle, B.H. et al. (1968): Trace metal accumulation by estuarine molluscs. J. Sanit. Eng. Div. Am. Soc. Civ. Eng. 94, 455-475.
3. Sims, R.R., Presley, B.J. (1976): Heavy metal concentrations in organisms from an actively dredged Texas Bay. Bull. Environ. Contam. Toxicol. 16, 520-526.

4. Watling, H.R., Watling, R.J. (1976): Trace metals in oysters from Knysna Estuary. Mar. Pollut. Bull. 7, 45-48.
5. Hernández, J.M., González, H., Isaac, M. (1987): Metales pesados en algunos moluscos de la Bahia de Manati, Cuba. Congreso de Ciencias del Mar, La Habana, Cuba.
6. Bryan, G.W., Langston, W.J., Hummerstone, L.G. (1980): The use of Biological Indicators of heavy metal contamination in estuaries. Mar. Biol. Ass. of U.K. Occasional publication No. 1.
7. Brooks, R.R., Rumsby, M.G. (1965): The biogeochemistry of trace element uptake by some New Zealand bivalves. Limnol. Oceanogr. 10, 521-527.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

HEAVY METALS IN FRESHWATER ORGANISMS IN THE CATCHMENT AREA OF LAKE BALATON

K.V.-BALOGH, J. SALÁNKI, I. VARANKA

Balaton Limnological Research Institute of the Hungarian Academy of Sciences, H-8237 Tihany, Hungary

ABSTRACT

Biological indicator organisms were used to quantitate heavy metal pollutants in the catchment area of lake Balaton. The species were Gammarus roeselii Gervais (crustacea), Lymnaea stagnalis L. (snail), Alburnus alburnus L., Carassius auratus gibelio Bloch (fish), which could be collected in most places within the catchment area.

We measured the concentrations of toxic heavy metals (Ni, Cu, Zn, Cd and Hg) in these organisms which live in various tributaries of the lake, by atomic absorption spectrophotometry. Animals were collected at 12 different locations altogether. The measurements were made during autumn in 1986 and 1987.

It was found that the concentrations of the studied metals in most of the biomonitor organisms are good means for signalling local pollution events. However, it seems that the muscle tissue of fish is not a suitable biomonitor object.

INTRODUCTION

Depending on the degree of environmental pollution, various harmful substances, among others heavy metals, can be accumulated in the body of aquatic organisms (Förstner and Prosi 1979, May and McKinney 1981). The potential of aquatic organisms to concentrate heavy metals has promoted their use as monitors of metal concentrations in marine and freshwater environments (Phillips 1980).

Most of the heavy metal pollution in Lake Balaton originates in the catchment area and enters the lake via the Zala River and other, smaller tributaries (Joó 1986, V.-Balogh and Salánki 1987).

The aim of the present investigation is to reveal whether the Zala River as well as various northern streams are polluted evenly by heavy metals or whether there are differences in the metal pollution they bring into the lake. This may refer to point-sources of pollution in the catchment region.

We compared the concentrations of heavy metals (Ni, Cu, Zn, Cd and Hg) in the biomonitor organisms, namely *Gammarus roeselii* Gervais (crustacea), *Lymnaea stagnalis* L. (snail), *Alburnus alburnus* L. and *Carassius auratus gibelio* Bloch (fish), which were collected in various tributaries of Lake Balaton. These species were chosen because of their ubiquity within the catchment area.

MATERIALS AND METHODS

The studies were carried out in the catchment area of Lake Balaton at the following study sites: site 1. Zala River (upstream from the town Zalaegerszeg); site 2. Zala River (downstream from the town Zalaegerszeg); site 3. Zala River (upstream from the Kis-Balaton reservoir); site 4. Kis-Balaton reservoir, near its inlet; site 5. Kis-Balaton reservoir, near its outlet; site 6. Héviz-Páhok canal; site 7. mouth of the Zala River; site 8. Edericsi brook; site 9. Lesence brook; site 10. Tapolca brook; site 11. Egerviz brook; site 12. Örvényesi brook. The animals were collected during autumn in 1986 and 1987.

Heavy metal concentrations were determined for the muscle tissue of fish and for the whole soft part of snails. The samples of *G. roeselii* included 20-45 individuals. Atomic absorption spectrophotometry was used as the analytical method. The samples were prepared by wet digestion for Hg analysis according to Paus (1972) and for other metals according to

Krishnamurty et al. (1976). Mercury was measured by the cold vapour method (Hatch and Ott 1968) utilizing the Spektromom Hg-detection unit of the Zeiss AAS1 type AA spectrophotometer. The other metal concentrations were measured in an air-acetylene flame using the same type AAS equipment. The data are related to dry weight. For mercury, the obtained concentration values were converted into dry weight by making use of the wet weight/dry weight ratios determined in parallel in every case. Considering all sampling sites and different species, these ratios fell into the range of 4.07-12.1. The concentrations given in the figures are mean values ($\pm$ standard error of mean, n=3).

RESULTS

Metal concentrations in Lymnaea stagnalis L.

In most cases of the studied heavy metals, significantly different values were obtained in the snails collected from seven out of twelve sampling sites (Fig. 1). The concentration of nickel in the snails was found to be below the detection limit at the sixth site. The highest Ni concentration (14.2 mg kg^{-1}) was found at the tenth site. Wide variation was detected with respect to copper concentration. The lowest one (13.5 ± 5.97 mg kg^{-1}) was measured at the fifth, and the highest (61.4 mg kg^{-1}) at the ninth site. The zinc concentration was around 70 mg kg^{-1} in the snails from most of the studied sites. However, a lower Zn concentration (39.4 mg kg^{-1}) was obtained at the tenth site. The cadmium concentration in the L. stagnalis individuals from the third site was the highest one (8.33 mg kg^{-1}), and it decreased to 3.14 ± 0.989 mg kg^{-1} at the reservoir (site 4), then increased gradually toward the mouth of the Zala River (7.22 ± 2.89 mg kg^{-1}, site 7). The lowest Cd concentration (2.19 mg kg^{-1}) was measured at the tenth sampling site. The distribution of mercury concentrations was opposite to those of cadmium, it was highest (0.684 mg kg^{-1}) at the Kis-Balaton reservoir (site 4), then decreased toward the mouth of the Zala River (site 7: 0.114 mg kg^{-1}).

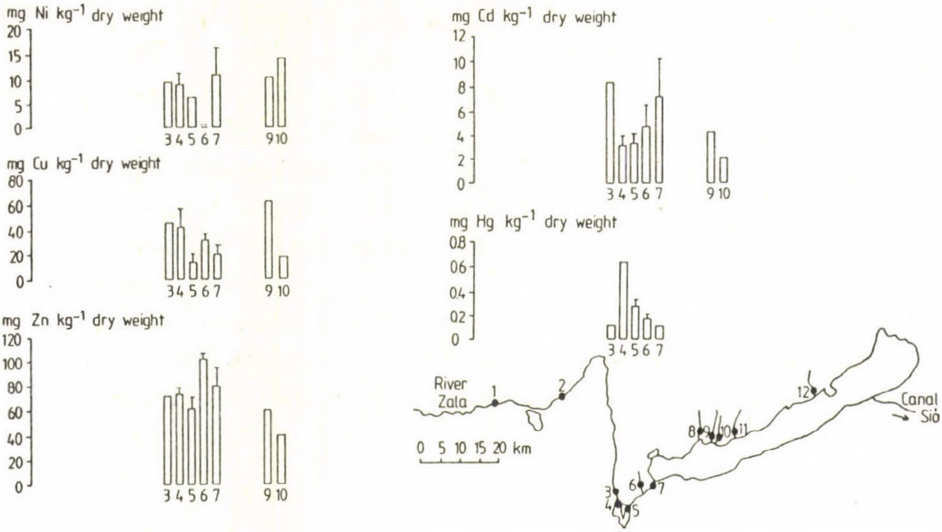


Fig. 1. Nickel, copper, zinc, cadmium and mercury concentrations in the soft body of Lymnaea stagnalis L. at the various biotopes.

Metal concentrations in Gammarus roeselii Gervais

We compared the heavy metal concentrations of G. roeselii at six sampling sites (Fig. 2). The concentration of nickel was found to be below the detection limit at the second site. The measurable nickel values fell into the range of 8.38 ± 1.14 mg kg^{-1} - 36.4 ± 2.65 mg kg^{-1}. The latter high Ni concentration was found at the eleventh site. The copper and zinc concentrations in the G. roeselii from the six sampling sites were around 60-80 mg kg^{-1}, without significant deviation. The lowest cadmium concentration (4.65 ± 1.65 mg kg^{-1}) was measured at the third site, while the highest (11.2 ± 1.06 mg kg^{-1}) was at the second site, which was on the Zala River downstream from the town of Zalaegerszeg. Wide variation was detected with respect to mercury concentration: the range of measurable values was between 0.156 ± 0.02 mg kg^{-1} and 1.38 ± 0.377 mg kg^{-1} at the third and eleventh sites, respectively. The concentration of Hg was found to be below the detection limit at the first site, which was on the Zala River upstream from the town of Zalaegerszeg.

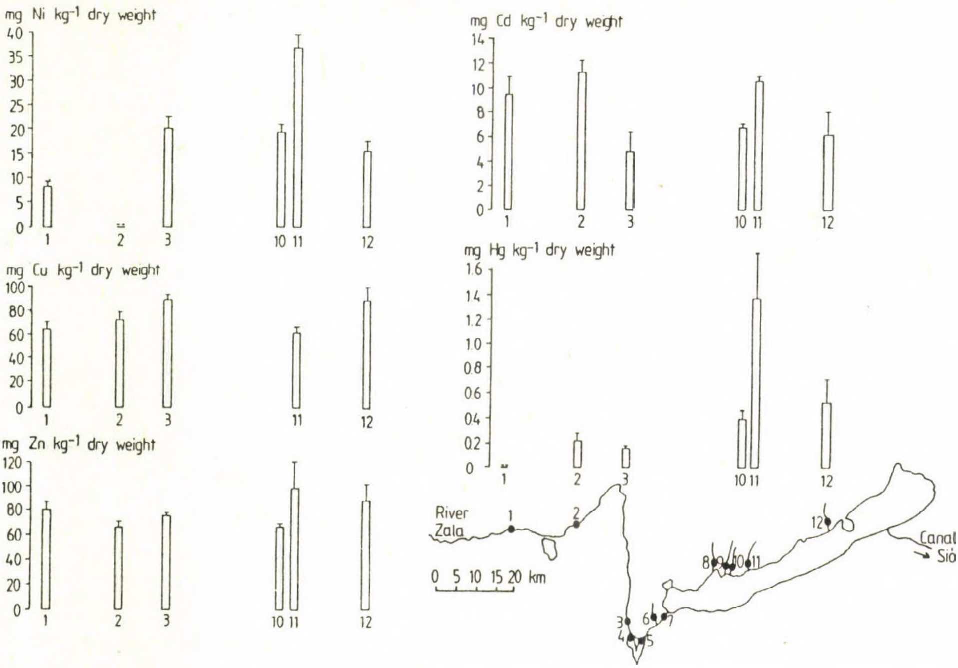


Fig. 2. Nickel, copper, zinc, cadmium and mercury concentrations in Gammarus roeselii Gervais at the various biotopes.

Metal concentrations in the fish species

The C. auratus samples were from sites 3, 4 and 6, while the A. alburnus samples came from sites 6, 8 and 10 (Fig. 3). The common sampling site was the sixth. The concentration of all studied metals showed significant differences in the muscle tissues of fish at this site. Nickel, copper and zinc concentrations (14.4 ± 1.06 mg kg^{-1}, 20.8 ± 2.65 mg kg^{-1}, 279 ± 54.8 mg kg^{-1}, respectively) were significantly higher ($P<0.01$) in the A. alburnus, while the concentrations of cadmium and mercury (3.54 ± 1.26 mg kg^{-1} and 1.40 ± 0.494 mg kg^{-1}, resp.) were higher in the muscle of C. auratus. Taking into consideration the values obtained in A. alburnus it seems that the concentrations of Ni, and Hg (about 20 mg kg^{-1} and 0.4 mg kg^{-1}, resp.) showed no significant differences at the three sampling sites. The concentration of Cd in this fish species was found to be below

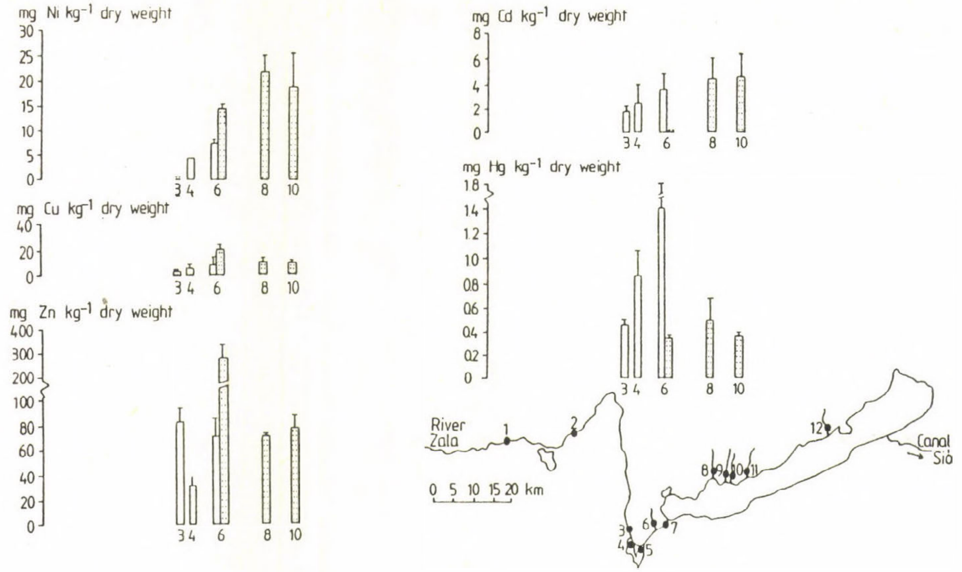


Fig. 3. Nickel, copper, zinc, cadmium and mercury concentrations in the muscle tissue of fish Carassius auratus gibelio Bloch (empty column) and Alburnus alburnus L. (spotted column) at the various biotopes.

detection limit at the sixth site. In case of Cu and Zn significant differences ($P<0.01$ and $P<0.001$, resp.) were observable between the fish collected from the sixth and two other sites. The heavy metal concentrations in the C. auratus changed in a similar direction. Except for zinc, an increasing tendency was observable in all other metals toward the sixth sampling site, where the concentrations were the following: Ni: 7.57 ± 0.625 mg kg^{-1}, Cu: 8.07 ± 5.60 mg kg^{-1}, Cd: 3.54 ± 1.26 mg kg^{-1} and Hg: 1.40 ± 0.494 mg kg^{-1}). The highest zinc concentration (83.9 ± 11.1 mg kg^{-1}) was found at the third location, while the lowest (33.9 ± 6.54 mg kg^{-1}) appeared at the inlet region of the Kis-Balaton reservoir (site 4).

DISCUSSION

Comparing the values obtained for L. stagnalis, it could be concluded that the Tapolca brook (site 10) was polluted with

Ni, the Lesence brook (site 9) with Cu, the Zala River (site 3 and site 7) with Cd, and the Kis-Balaton reservoir (site 4) with Hg. It was observable that the Hg concentration in snails decreased gradually after the reservoir. In our earlier study (V.-Balogh et al. 1988) it was shown that L. stagnalis was a suitable organism for monitoring heavy metal pollution in the environs of Lake Balaton. It is important to keep in mind that these earlier results also showed a Hg-pollution at the Kis-Balaton reservoir.

Based on the heavy metal concentration values of the G. roeselii samples it was obvious that the Egerviz brook (site 11) was polluted with Ni, Cd and especially Hg. The reach of the Zala River upstream from the town of Zalaegerszeg (site 1) was not polluted with Hg.

By studying the heavy metal concentrations of the muscle tissue of the two fish species it was found that they accumulated metals in different degrees independent of the studied sites. Significantly higher concentrations of Ni, Cu and Zn were found in A. alburnus while the Cd and Hg concentrations were considerably higher in C. auratus. In an earlier work (Salánki et al. 1988) we also conducted measurements with two fish species, A. brama L. and Stizostedion lucioperca L. at Lake Balaton. There were differences in the metal uptake of various organs between the two species even when the actual water concentrations seemed similar. These differences may have a connection with the feeding habit of the various fish species.

The C. auratus of the two fish species is rather suitable as monitor organism on the basis of its higher body weight. The data on the metal concentrations detected in muscle tissue of C. auratus suggests the heavy metal loading of the Héviz-Páhok canal (site 6) for most of the studied five metals. However, this observation was supported by results for A. alburnus in cases of Cu and Zn only.

The results seem to justify the supposition that the used biomonitor organisms are good means for signalling local pollution events. However, it seems that the muscle tissue of fish is not a suitable biomonitor. On basis of the monitor organisms, it can be concluded that the 12 study sites in the catchment area of Lake Balaton are nonuniformly polluted with heavy metals.

REFERENCES

Förstner, U. and Prosi, F. (1979) Heavy metal pollution in freshwater ecosystems. In: Biological Aspects of Freshwater Pollution (ed. by Ravera, O.). Pergamon Press, Oxford-New York, pp. 129-161.

Hatch, W.R. and Ott, W.L. (1968) Determination of submicrogram quantities of mercury by atomic absorption spectroscopy. Analyt. Chem. 40, 2085-2087.

Joó, O. (1986) Role of the Zala River in the eutrophication of Lake Balaton. In: Modeling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton (ed. by Somlyódy, L. and van Straten, G.). Springer Verlag, Berlin-Heidelberg-New York-Tokyo, pp. 341-356.

Krishnamurty, K.V., Shpirt, E. and Reddy, M.M. (1976) Trace metal extraction of soils and sediments by nitric acid-hydrogen peroxide. Atom. Absorp. Newslett. 15, 68-70.

May, T.W. and McKinney, G.L. (1981) Cadmium, lead, mercury, arsenic and selenium concentrations in freshwater fish, 1976-77 National Pesticide Monitoring Program. Pesticides Monitoring Journal 15, 14-38.

Paus, P.E. (1972) Bomb decomposition of biological materials. Atom. Absorp. Newslett. 11, 129-130.

Phillips, D.J.H. (1980) Quantitative aquatic biological indicators. Pollution Monitoring Series (Adv. ed. by Mellanby, K.). Applied Science Publishers Ltd., London, pp. 1-488.

Salánki, J., V.-Balogh, K. and Hernádi, L. (1988) Biomonitoring of the state of the environment with reference to heavy metal pollution of fish in Lake Balaton. In: Biological Monitoring of Environmental Pollution (ed. by Yasuno, M. and Whitton, B.A.). Tokai University Press, pp. 55-60.

V.-Balogh, K. and Salánki, J. (1987) Biological monitoring of heavy metal pollution in the region of Lake Balaton (Hungary). Acta Biol. Acad. Sci. Hung. 38, 13-30.

V.-Balogh, K., Salabarria Fernandez, D.M., Salánki, J. (1988) Heavy metal concentrations of *Lymnaea stagnalis* L. in the environs of Lake Balaton (Hungary). Water Res. 22, 1205-1210.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

WATER QUALITY AT INLETS AND OUTLETS OF SOME FISH PONDS IN THE WIELKOPOLSKA REGION

A. KOSTURKIEWICZ, S. MURATOWA, A. MILER

Katedra Melioracji Rolnych i Leśnych, Agricultural University
ul. Wojska Polskiego 71e, 60-625 Poznań, Poland

1. INTRODUCTION

The purpose of the present investigations was to analyse water quality at fish ponds inlets against a background of fish-culture requirements and its changes after using water in ponds. Three fish ponds in the Wielkopolska region (Poland) were chosen for the investigations. Mainly carps (Carpinus carpio) and some tench (Tinca vulgaris) and amur (Ctenopharyngodon idella) were raised in the ponds. The main characteristics of the ponds and their feeder streams are given in Table 1.

The feeder streams were polluted by waste waters and agricultural non-point sources. Since May 1986 to September 1988 water quality was analysed at the inlets and outlets of the ponds once a month. The following quality characteristics were taken into account: dissolved oxygen (DO), biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), temperature, pH, solids, ammonia, nitrates, nitrites, orthophosphates, hardness, sulphates, chlorides, calcium, magnesium and potassium. This year (1988) additionally some heavy metals (Pb, Zn, Cd) were determined twice (in July and September) and in July a hydrobiological survey was accomplished. Besides, total iron concentration was analysed once a month.

The results of the above-mentioned analysis are briefly described below.*

---

*The investigations were carried out within the framework of Research Program RR-II-20 financially supported by the Ministry of National Education.

Table 1. Main characteristics of the investigated fish ponds and their feeder streams

PONDS				
Location		Dabrówka	Gorzyn	Stroszki
Denotation		D	G	S
Water surface area	ha	3.9	0.8	2.8
Mean depth	m	1.3	0.9	1.1
Mean inflow	dm^3/s	9.0	4.2	10.8
Mean outflow	dm^3/s	0.9	1.4	3.9
Hydraulic residence time	days	65	20	33
FEEDER STREAMS				
Name	-	Dabrówka	Struga Dor.	Maskawa
Mean streamflow	dm^3/s	22	164	246
Watershed area	km^2	7.3	44.5	37.2

2. QUALITY OF WATER IN THE PONDS

2.1. Pond D

Because of the spring origin of the inflowing water, its temperature (Fig. 1) was relatively low: the maximum value observed on 4th of June 1988 was 19.5°C. Water temperature at the outlet was as a rule a few centigrades higher (up to 8°C in August 1988) than at the inlet, except at the beginning of the farming season (e.g. March 1988). Even the lowest concentration of DO in the pond water observed in August 1988 (6.4 mg O_2/l) was high enough for normal fish growth. It is alarming that the five-day BOD_5 at the end of the last farming season exceeded more than twice the mean-time BOD_5 value.

The hydrobiological surveys did not indicate algal blooms in the pond. The pH both at the inlet and at the outlet showed an increasing trend (Fig. 2) with a positive impact on fish development, the more so as the ammonia concentrations were relatively low.

A negative impact on fishes was caused by the increase in the concentration of total iron which at the end of last summer exceeded the safe value, being 0.9 mg Fe/l according to Starmach et al. (1976). High levels of iron in the inflowing

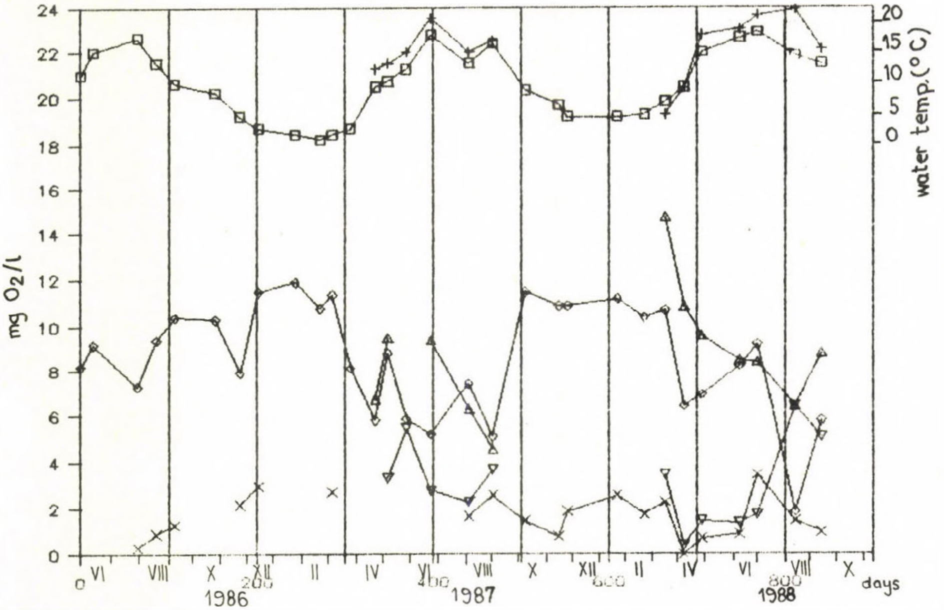


Fig. 1. Pond D: water temperature (□ inlet, + outlet), DO (◇ inlet, △ outlet) and BOD_5 (x inlet, ▽ outlet)

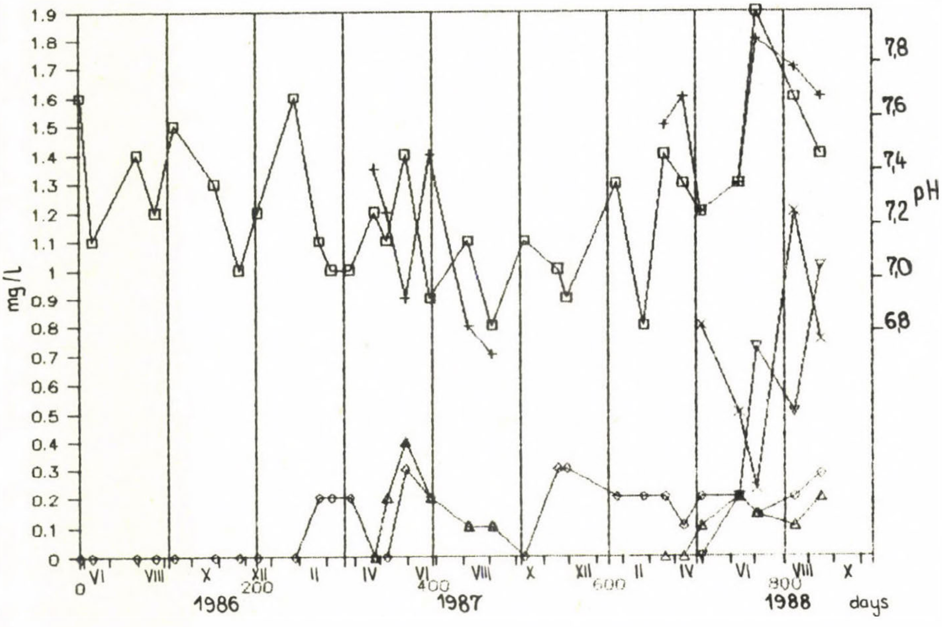


Fig. 2. Pond D: pH (□ inlet, + outlet), ammonia (◇ inlet, △ outlet) and total iron (x inlet, ▽ outlet)

waters were probably connected with their spring (underground) origin.

The most important nutrients in this case were orthophosphates, which occasionally occurred at high concentrations (≤ 0.5 mg PO_4/l). The nitrate nitrogen content in the feeder stream in the last two seasons was distinctly lower than in 1986 (Fig. 3). Also the nitrite nitrogen content (≤ 0.14 mg $N\text{-}NO_2/l$) was relatively small and beneficial to fishes.

The third important nutrient, potassium, remained all the time around the mean level of a few mg K/l (Fig. 4). Concentrations of calcium were constantly high (on the average 120 mg Ca/l). It had a positive influence on primary production and pH.

Most often the concentrations of calcium at the outlet were less than at the inlet, as a result of calcium deposition during photosynthesis.

A similar process was observed for magnesium, too. Concentration of heavy metals (Pb, Zn, Cd) did not reach levels harmful for algae and fishes.

2.2. Pond G

The temperature of water in the pond was all the time higher than at its inlet. That effect of water warming in the pond increased gradually during each farming season. However, despite the fact that the pond water was less warm that summer than a year before, the concentrations of DO were lower, showing a decreasing trend down to 4 mg O_2/l (Fig. 5). At the same time, DO levels in the feeder stream fluctuated around 6.5 mg O_2/l, and BOD_5 did not exceed 4 mg O_2/l. The main cause of the lack of DO in the pond was the introduction of $\simeq 2 \cdot 10^5$ of carp fry in the spring, i.e. almost twice the required quantity. The second cause was the feeding of fry with a liquid manure (two times by $\simeq 8$ m^3). Additionally, pond G cannot be fully drained out of the farming seasons and its bottom consists basically of organic sediments. Under these conditions the bottom was, beside the inflowing waters, one of the main sources of phosphates the concentrations of which were relatively high (Fig. 6). Though nitrate concentration in summertime was rather low,

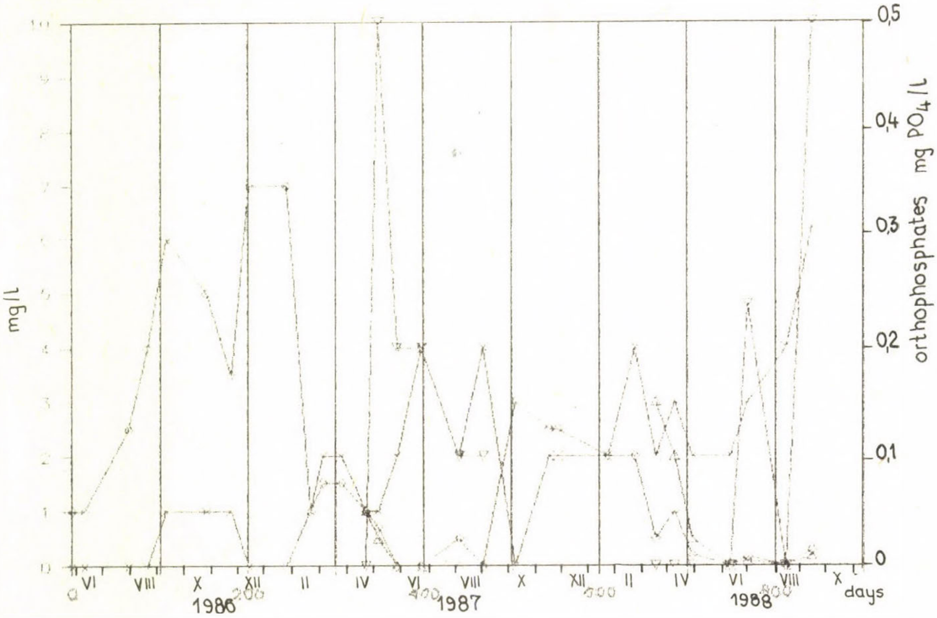


Fig. 3. Pond D: nitrates (◇ inlet, △ outlet) and orthophosphates (x inlet, ▽ outlet)

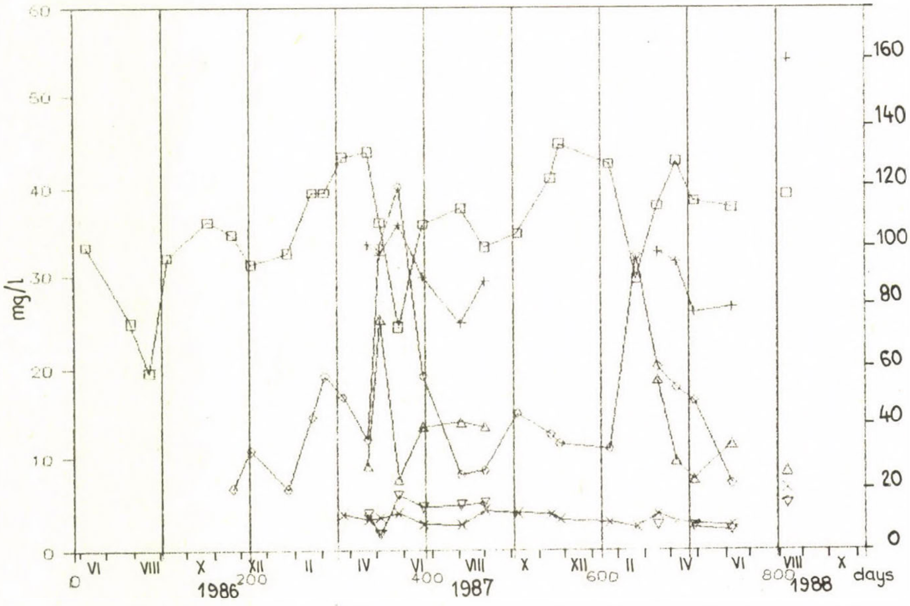


Fig. 4. Pond D: calcium (□ inlet, + outlet), magnesium (◇ inlet, △ outlet) and potassium (x inlet, ▽ outlet)

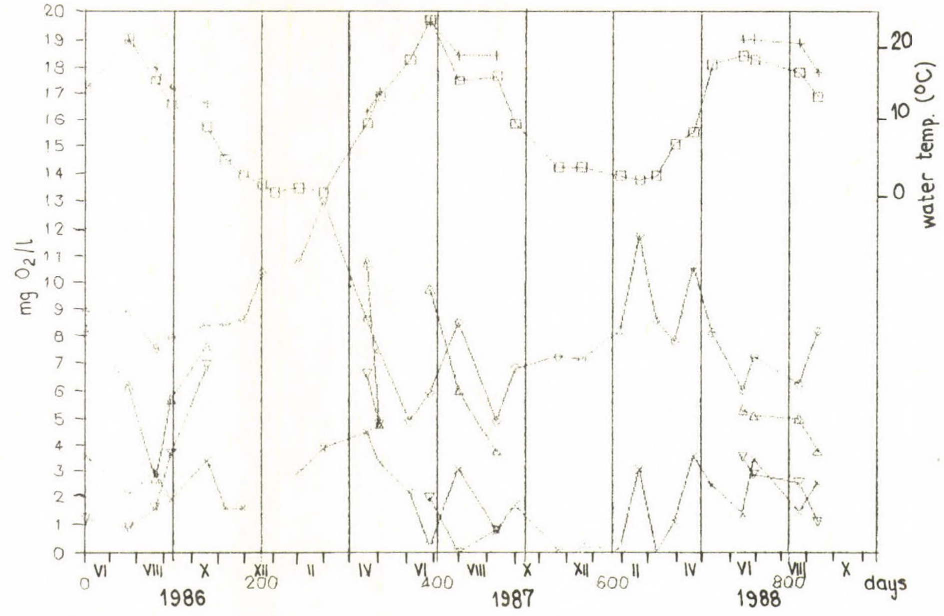


Fig. 5. Pond G: water temperature (□ inlet, + outlet), DO (◇ inlet, Δ outlet) and BOD_5 (x inlet, ∇ outlet)

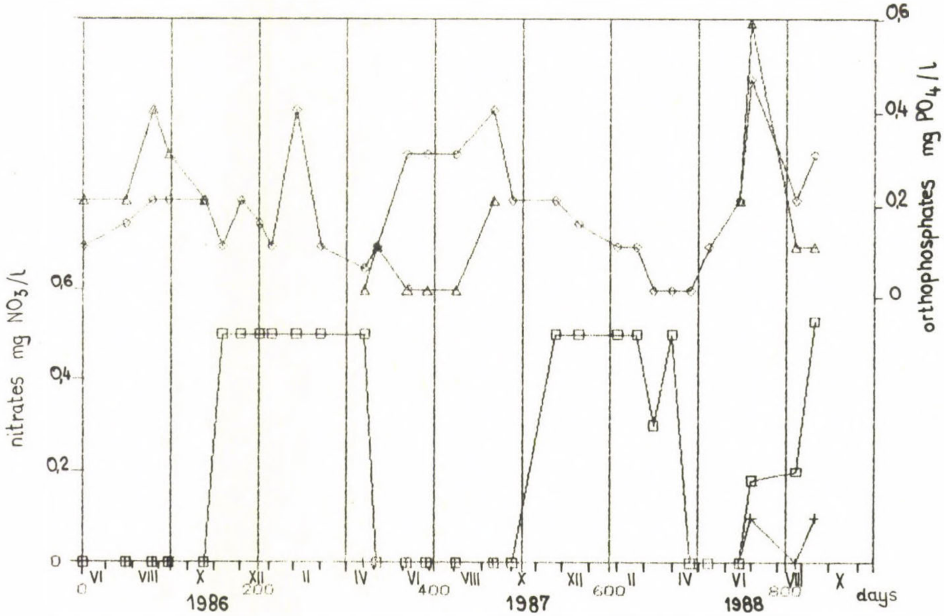


Fig. 6. Pond G: nitrates (□ inlet, + outlet) and orthophosphates (◇ inlet, Δ outlet)

an intensive growth of rooted macrophytes with floating leaves and phytoplankton was still observed. The pH indicated a positive increasing trend, up to 8.0 (Fig. 7) most probably due to a similar increase in the concentrations of calcium in the inflowing water (Fig. 8). Unfortunately, a similar trend was observed this season for iron, with a concentration exceeding the safe value of 0.9 ÷ 1.0 mg Fe/l.

The analysis of heavy metals showed a significant increase in their concentrations up to levels considered harmful to fishes both in the feeder stream and in the pond (Pb - 0.10 mg/l, Zn - 0.21 ÷ 0.23 mg/l, Cd - 0.009 mg/l). It could be caused, among others, by the proximity of a busy road. It should be mentioned that the water in the pond was not too hard (≃5 mval/l) and under the conditions of DO deficiency, as observed by Krenkel and Novotny (1980), the toxicity of heavy metals might be much higher than in a hard and properly oxygenized water.

All the negative factors mentioned above resulted in a very high mortality rate of the carps (97%!) during the last studied season.

2.3. Pond S

DO content and BOD_5 in the inflowing water were relatively low during summertime and, apart from the warm months, they were about two times higher (Fig. 9). The DO concentrations in the pond water were by 1-7 mg O_2/l higher than in the inflowing water and their mean values rose up to 10.2 mg O_2/l in 1988. Similarly, an increasing trend of the pH up to 8.0 was observed (Fig. 10). The distinct increase in ammonia and iron concentrations at the end of the last farming season was most probably caused by a rapid decrease in temperature and associated mixing processes.

In 1986 and 1988 inflowing waters were rich in orthophosphates and their concentrations at the outlet were several times less (Fig. 11). Only in 1987 were both concentrations practically equal.

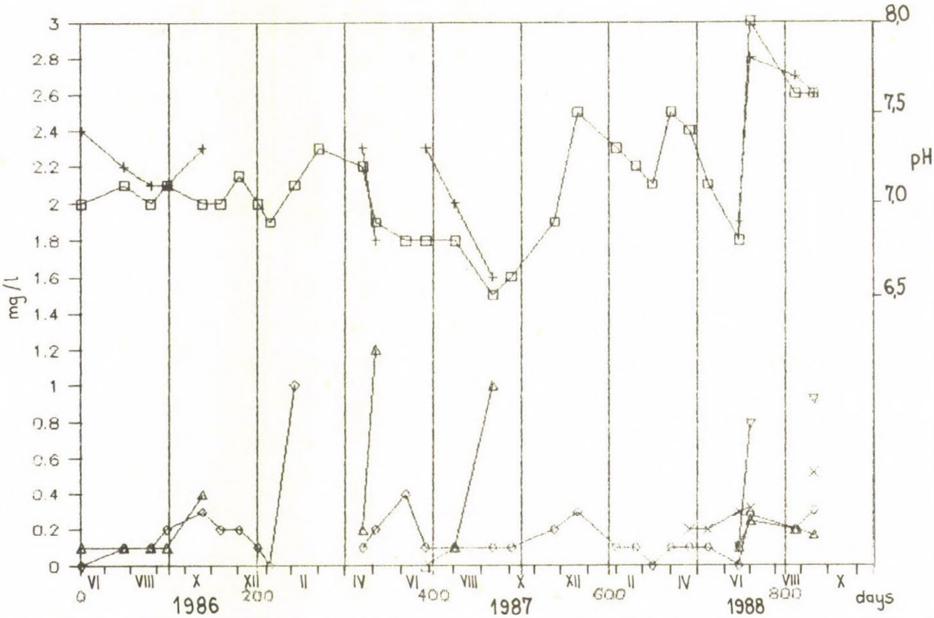


Fig. 7. Pond G: pH (□ inlet, + outlet), ammonia (◇ inlet, △ outlet) and total iron (x inlet, ▽ outlet)

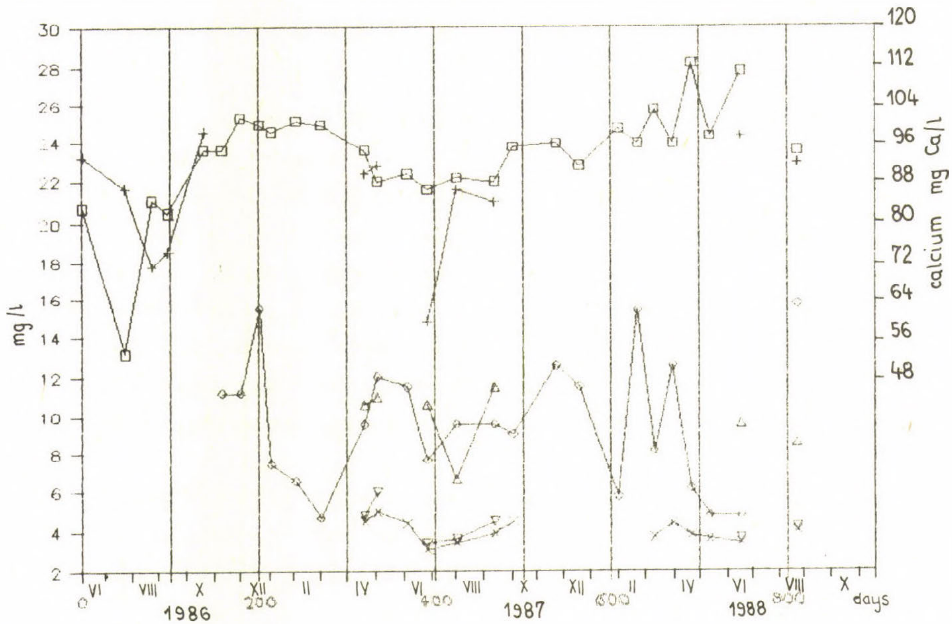


Fig. 8. Pond G: calcium (□ inlet, + outlet), magnesium (◇ inlet, △ outlet) and potassium (x inlet, ▽ outlet)

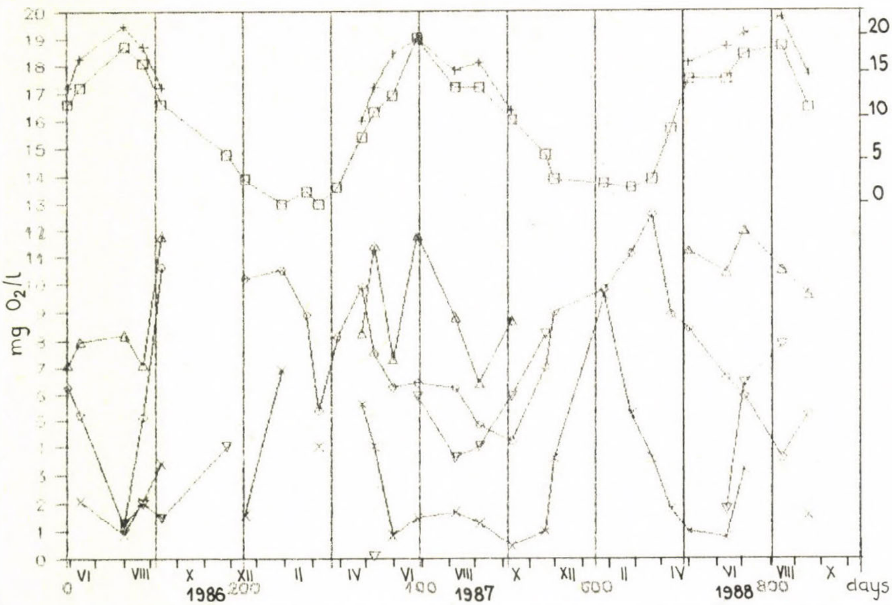


Fig. 9. Pond S: water temperature (□ inlet, + outlet), DO (◇ inlet, △ outlet) and BOD_5 (x inlet, ▽ outlet)

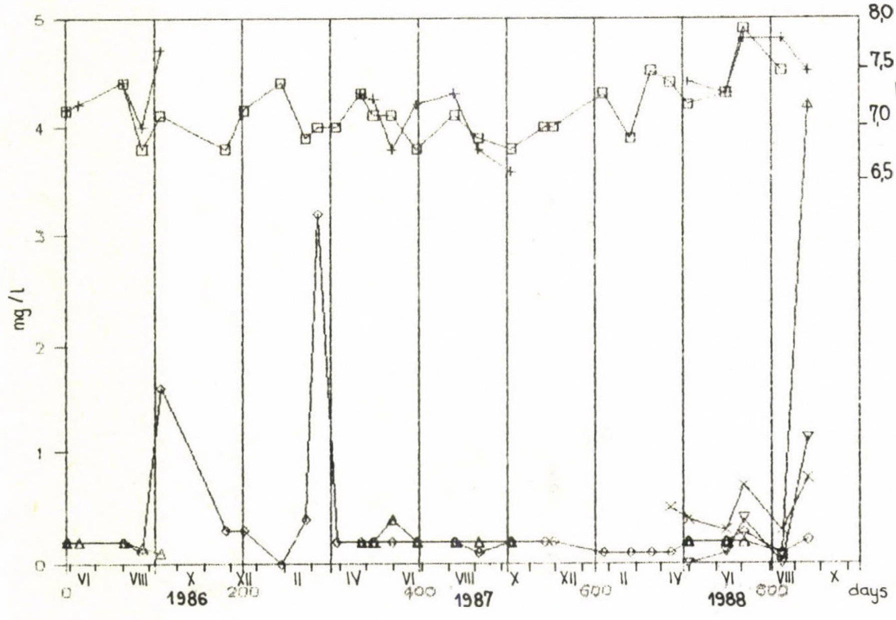


Fig. 10. Pond S: pH (□ inlet, + outlet), ammonia (◇ inlet, △ outlet) and total iron (x inlet, ▽ outlet)

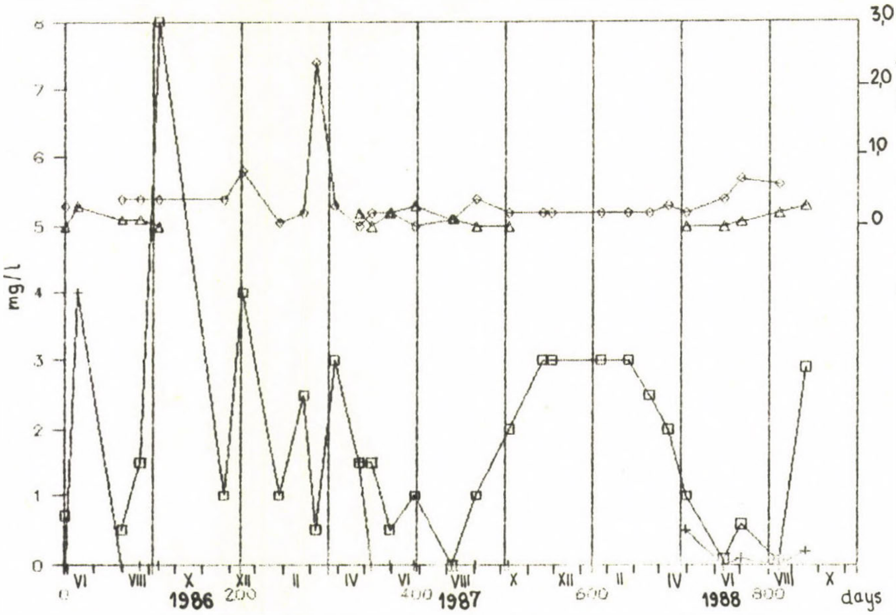


Fig. 11. Pond S: nitrates (□ inlet, + outlet) and orthophosphates (◇ inlet, Δ outlet)

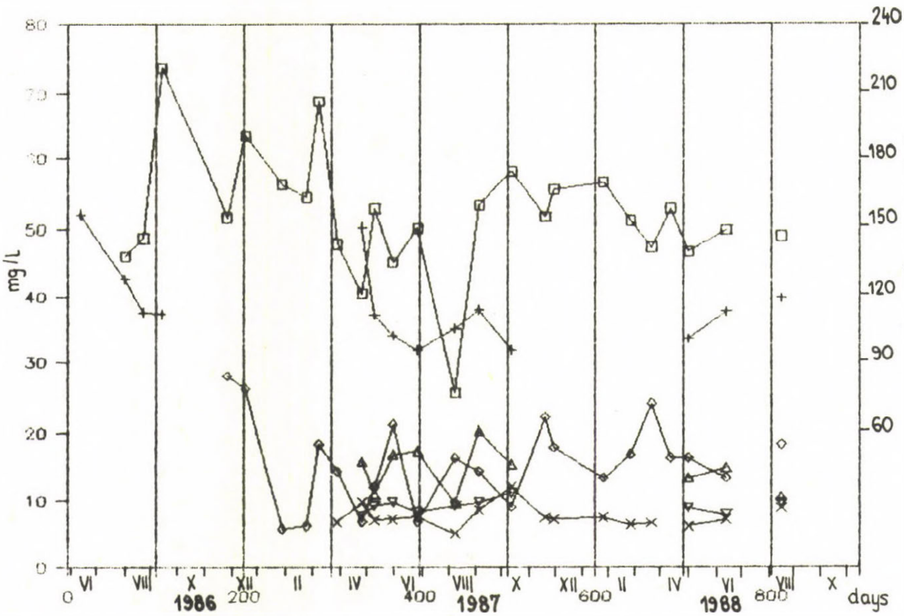


Fig. 12. Pond S: calcium (□ inlet, + outlet), magnesium (◇ inlet, Δ outlet) and potassium (x inlet, ∇ outlet)

The chemical analysis did not show any significant increase in calcium, magnesium and potassium concentrations which remained at relatively high levels (Fig. 12).

Sediment samples consisted of medium sand with $\simeq$1% of humus and 3-4% of CaO. The latter high value was due to an artificial liming of the pond. High concentrations of heavy metals in the sediments and much lower ones in the water at the outlet than at the inlet showed that at least cadmium and lead were cumulated in the sediment and very likely in the fish tissues.

3. CONCLUSIONS

Fish ponds are not simply sedimentation tanks but they enable a tertiary treatment, especially in cases of heavily polluted inflowing waters.

A complete draining of fish ponds out of the farming season is desirable for it accelerates oxidation and decomposition of organic sediment and improves the productivity of the ponds. Preliminary results of the investigations periodically show dangerous increases in concentrations of heavy metals.

REFERENCES

Krenkel, P.A., Novotny, V. (1980): Water Quality Management. Academic Press, New York.

Starmach, P.A., Wróbel, S., Pasternak, K. (1976): Hydrobiologia. PWN, Warszawa.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

CHANGES INDUCED BY ANTHROPOGENIC POLLUTION IN THE WATER RESOURCES OF NORTHERN KOLA

T.I. MOISEENKO

The Kola Scientific Centre of the USSR Academy of Sciences
Apatity, USSR

Northern Kola is one of the most developed industrial regions of the Soviet Union. The environment, particularly the large reservoirs Imandra, Umbozero, Lovozero have been polluted for more than fifty years. A rapid deterioration of the aquatic ecosystems has taken place in the region due to the increased external loadings of different origin (industrial, agricultural, communal, etc.).

Sewages of the metallurgic and mining plants contain mineral salts, suspended solids, heavy metals, residuals of chemicals used for flotation, fluorides, etc. in high concentrations. According to their ecological effects, two groups of chemicals can be distinguished: 1. Chemicals which influence aquatic organisms directly and have toxic effects in the reservoirs, i.e. heavy metals and organic compounds used for flotation of apatite-nepheline ores as well as for producing nonferrous metals. 2. Chemicals which change the physicochemical parameters of the aquatic ecosystems. Entering the reservoirs, these materials increase mineralization, decrease the transparency of water and change its ionic composition, etc.

Sulphur-containing compounds and metals emitted by nonferrous factories reach the watershed areas of the water bodies with rain and snow. Metals spread over in the atmosphere intoxicate even reservoirs which are far away from the industrial zone and are not polluted directly. The atmospheric fall-out of sulphur-containing compounds has serious ecological side-effects. In the Murmansk region the estimated fall-out of sulphur-containing compounds exceeds 4.8 tons/km^{-2} (Makarova et al. 1984, Bryukhanov et al. 1982). In this region the pH decreased in a number of reservoirs and water courses (Izrael et al. 1983). Fortunately, in the central part of the Kola Peninsula the buffer capacity of the watershed basins is considerably high due to the alkaline rocks of the Khibini and Lozorevo Mountains. In this region acid precipitation leads to an increased washout of sulphates and toxic metals, particularly aluminium from the rocks.

Thermal pollution of the reservoirs is a consequence of the heating water discharge (8-12°C) from nuclear power plants. Increased sewage loading of oligotrophic water bodies induces eutrophication. The sewage originates mainly from industrial and municipal sources as well as from cattle breeding.

The ecological side-effects of industrial development can clearly be illustrated by the example of Lake Imandra. The hydrochemical regime of the lake has changed due to the increased pollution: mineralization increased by a factor of 3-10, the bicarbonate-type water changed into a sulphate-type water, transparency decreased due to the enhanced loading of fine-grained suspended matter. Concentrations of heavy metals and toxic organic compounds increased in the water, similar to the concentrations of the nutrients. Eutrophication was intensified by the thermal pollution. Toxicity of metals was also higher in the warmer water.

Similar hydrochemical changes were observed also in Lakes Lovozero and Umbozero: rates of mineralization as well as concentrations of suspended solids, sulphate, toxic compounds and nutrients increased. Concentrations of heavy metals increased in a number of other reservoirs (Lakes Pirenga, Chunlake, etc.) and tributaries (Varzuga, Kolvitsa, etc.), too, as a result of atmospheric pollution of their watershed areas.

Ten years' research of the lakes in Northern Kola revealed that diversity of aquatic organisms decreased in response to the changes of the aquatic environment (see Table 1). The typical organisms inhabiting the northern reservoirs are very sensitive to pollution. Species characteristic of cold water, oligotrophic reservoirs are outnumbered by euribiotic and cosmopolite species. A striking example of such a change is Lake

Table 1. Changes of some parameters in the lakes of Northern Kola

Parameter	Natural background (1)	Origin of pollution				
		Metallurgy (2)	Apatite-nepheline and communal (3)	AES heating water (4)	Mining (5)	Pollution from watershed (6)
Total salts, mg l^{-1}	17-24	90-210	95-100	40-60	30-80	30-40
Sulphate, mg l^{-1}	3	29-105	28-43	14-20	3-6	5-14
Susp. solids, mg l^{-1}	0.4-0.7	2-30	2-45	1-3	4-25	0.7-1
Heavy metals, conv. units	1-3	30-200	3-10	1-5	1-5	2-10
Toxic org. compounds (mg l^{-1})	-	0-1	0-0.7	-	-	-
Fluoride, mg l^{-1}	0.15	0.15	0.1-1	0.15	1-8	0.15
Total P, $\mu g\ l^{-1}$	1	10-110	10-800	1-24	2-200	5-7
Inorganic N, $\mu g\ l^{-1}$	1	10-80	10-3800	1-5200	2-500	20-80
Biomass of the benthos (g m^{-2})*	1-1.2	0-50	9-300	1-3	7-12	1-1.2
Diversity, bit	2.5	0-2.5	0.2-2.5	1-3.5	1-3.5	1-3.5
Number of fish species	14	2	3	11	14	14

1 - Lakes Imandra (16 km^2), Lovozero and Umbozero (100 km^2); 2 - Lake Imandra; 3 - Lake Imandra, 4 - Lake Imandra, t = 3-8°C; 5 - Lakes Umbozero and Lovozero; 6 - Lake Pirenga (160 km^2)

*Yakovlev (1981)

Imandra receiving large amounts of industrial and communal pollutants. The plankton is represented by a few species only, and the planktonic production is low. Chironomids dominate the zoobenthos with a biomass up to 50 g/m^{-2} near the inflows in which concentrations of copper and nickel are high, whereas tubificides are dominant (biomass being maximum 200 g/m^{-2}) in the areas where the tributaries are of apatite-nepheline nature (Yakovlev 1982).

Fish are exposed both directly and indirectly to pollution. The direct effects include toxicity of the pollutants, whereas indirect ones decreased biomass of the food organisms, primarily zoobenthos. These effects result in migration of fish from such areas. Acute toxic exposure of fish arriving from the tributaries into the lakes leads to their perishing. The fish stock decreases even if the area of toxic zones is small in the lakes.

Pollutants can even be detected in significant tributaries in low concentrations. Changes were observed in the fish population (Coregonus lavaretus, C. albula, Salventus lepechini) of the polluted Lake Imandra, which, however, has not received direct industrial loading for more than 20 years. Due to the chronic intoxication resulting in different diseases and dysfunctions, the mortality of fish increased, their growth rates decreased. The length and weight of Coregonus lavaretus and C. lepechini are typicallly reduced in each age group in populations of the polluted areas in comparison to the control populations of non-polluted areas. Simultaneously with the decrease in the growth rate of these fish, fat accumulation showed an increase. Spawning is delayed.

Thermal pollution of the northern lakes has its peculiarities. The heated water spreads over a 2-3 m surface layer. Convective currents are generated at the bottom. Thermal pollution of the sediments and benthic communities, however, is negligible in comparison to that of the surface layers. First of all the cold-adapted, relict species disappear. The species diversity decreases, similar to that observed during eutrophication. At the same time, the length of the vegetative season increases. This results in an increased productivity in the areas exposed to thermal pollution. Mortality of the cancroids may reach 70%, and the remaining population suffers from heat stress (Moiseenko et al. 1988). The temperature of the thermally polluted subarctic lakes does not exceed the physiological optima of salmon and sig species (18°C) only for a short summer period (end of July to early August) in some places where the water temperature may reach 24°C. In this period fish migrate from these areas, their growth rate being 1.5 times less than in areas where water temperature is close to the physiological optimum. Premature, asynchronous spawning can be observed in fish of the highly polluted areas. Reproduction of fish is disturbed because the ripening of the spawn and the thermal conditions for spawning are out of phase. Therefore the spawn is exposed to unfavourable incubation conditions. Moreover, spawning areas are also degraded. These factors finally lead to a decrease in the fish stock of the lakes.

REFERENCES

Bryukhanov, P.A., Kryuchkov, V.V., Nasarov, I.M., Ryaposhapko, A.G. (1982): Estimation of concentrations of sulphur dioxide and sulphate transport in the USSR. IPG, 41, 14-21.

Izrael, Yu.A., Nazarov, I.M. (1983): Acid rains. Publ. House Hydromet. 207 pp.

Makarova, T.D., Artobolevsky, V.I. (1984): Monitoring of contaminating substances and atmospheric precipitation in the Northern Kola. Apatity, pp. 63-70.

Moiseenko, T.I., Yakovlev, V.A., Lukin, A.A. (1988): Use of the AES heating water in fishfarming. In: Industrial Effects in the Aquatic Ecosystems of Northern Kola. The Kola Branch of Acad. Sci. USSR, Apatity.

Yakovlev, A.V. (1981): Anthropogenic influences on the zoobenthos of an oligotrophic reservoir. In: State and Prognosis of Northern Kola, pp. 48-50.

LAKE PROTECTION OVER THE WORLD

AFRICA

ASIA

AMERICAS

EUROPE

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

HUMAN INTERVENTION IN NATURAL PROCESSES OF THE LAKE VICTORIA ECOSYSTEM. THE PROBLEM

F.W.B. BUGENYI, J.S. BALIRWA

Uganda Freshwater Fisheries Research Organisation
P.O. Box 343, Jinja, Uganda

ABSTRACT

Human activities in the Lake Victoria basin have interacted with natural processes and ultimately with the stability of the aquatic ecosystem. The functioning of this ecosystem is yet to be fully understood. It has complex biological, chemical and physical features which make it a particularly fragile ecosystem easily disrupted by human biomanipulation. The water resources from this ecosystem have always been of great economic and social importance to the three riparian states. Apart from providing food from the naturally rich diverse fish fauna, the people of the region benefit from the water resources through employment opportunities, water supplies for domestic, industrial and agricultural purposes, and, as a medium for transportation and recreation.

The lake faces imminent problems due to various factors. These include widespread deforestation and the consequent soil erosion in the lake watersheds; pollution from agricultural pesticides in the run-off, and cultural eutrophication from fertilizers and industrial effluents. It now appears that stocking with alien fish has contributed to the elimination of numerous endemic fish species, a related factor being the use of destructive fishing gears and methods. All these, compounded by weaknesses in application of effective management policies on one hand, and, environmental processes on the other, would also render effective management largely uncoordinated.

More interdisciplinary research is needed in order to understand the dynamics of ecosystem functioning from which the formulation of management measures that will ensure rational development and conservation will be made.

INTRODUCTION

Lake Victoria, one of the Great African Lakes, is the largest lake in Africa, and is next to Lake Superior in area among the World's Lakes (Livingstone, Melack 1984). It has a surface area of 68,635 km^2, 43% of which is in Uganda. It has a total water volume of 27,000 km^3 with a maximum depth of 80 m and mean depth of about 20 m. The Lake is ponded between shoulders of both the Western and the Gregory Rifts and has been locally deepened and extended by block faulting.

Rainfall, temperature, wind, sunshine and air circulation are some of the elements that make up the climate of a place. The two main wind systems affecting the climate of East Africa in general and Uganda in particular, are the north-east and south-west trade winds. The entire catchment area of L. Victoria is within the zone where the two wind systems meet. This zone receives heavy rains from conventional storms in two seasons related to the trade wind passage periods and equatorial equinoxes. The total amount of rain received and its distribution within the catchment throughout the year have a great bearing on the lake level. A very important determinant of the nature of any lake is the ratio of ponded water to river flow through it. In the humid equatorial regions much of both rainfall and evaporation impinges directly on the lake surface or the catchments of very short tributaries. The main water loss is by direct evaporation from the lake surface. Under such circumstances it is possible for increased run-off to the lake or reduced evaporation from it to result in a rise of several metres in water level, even though the lake possesses an outlet (Livingstone, Melack 1984). Lake level changes have great effects on the lake's fish productivity (Ssentongo, Welcomme 1985). Following the unusually heavy rains between 1961 and

1963, for example, there were improvements in the fish catches from Lake Victoria (Welcomme 1966).

Lake Victoria is located in one of the most densely populated areas of Eastern Africa, where it is therefore of crucial socioeconomic importance as a source of water (for domestic, industrial and agricultural uses), as a site of the important fisheries and of recreation and transport. Certain localities around the lake have formed foci for development where traditional ways of life and fishing have been largely superseded (Ribbink 1987).

The development of human societies, and of the cities, has always been conditioned by the availability of the water resources essential for their existence. And, despite all its technological transformations, the modern world has not changed this immutable law. On the contrary, the problem of water has, in our days, assumed new dimensions, due partly to the growth of the populations and the consequent greater need for water. The rapid increase in the needs of agriculture, industry and the phenomenon of urbanization has increased the demand for water resources.

The economy of the catchment of L. Victoria in Uganda is dominated by subsistence agriculture, where 90% of the population is involved in the production of food and cash crops. There is envisaged intensification of agriculture (Bugenyi 1984a) because of increased population and because of the need to increase food production. This has involved more use of agricultural pesticides and fertilizers, land clearing and deforestation (Hamilton 1984) for more acreage, all of which have been mismanaged. The agrochemicals and the eroded soil are transported down to the lake by rivers, streams and rainfall run-offs.

The urban centres along the catchment, mainly Kampala, Jinja, Masaka and Entebbe, have industries involved in textile manufacture, mining and smelting, food processing, beer brewing and soft drinks manufacture, etc. These industries drain their waste products and effluents into L. Victoria. These centres have sewage systems, which are let through to the lake waters.

The lake fisheries are heavily exploited, particularly in the vicinities of these centres.

The Lake is inhabited by at least 250 species of fish many of which show little stock mobility and are habitat restricted, while some undertake spawning migrations to rivers and streams. The fish are caught by fishermen using various methods and techniques which exploit knowledge of fish habitat and behaviour. Over the years an increase in fishing intensity and innovation have resulted in a decline in catches. Some of the fishing methods and gears have also been responsible for the decline.

There have also been ill-advised introductions in the fishery of a predatory fish, Nile perch and several herbivorous Tilapias (Fryer 1960, Jackson 1960, Okemwa 1984, Ogutu-Ohwayo 1985, Barel et al. 1985, Coulter et al. 1986). The effects of fishing and the introductions have been a change in stock abundance and species composition far different from the original fish communities in the lake. This has come at a time of increased demand for protein as a result of the increasing human population in the lake basin. With increased pressure on land through agricultural practices, extensive areas around the lake are cultivated with the result that erosion associated with mineral nutrients in the drainage basin could be causing a shift in the primary productivity of the lake. In future, industrial effluents and domestic discharges in the lake could become an additional nutrient source - thus hastening eutrophication especially in the absence of many herbivorous species and detritus feeders in the trophic structure of the lake.

Regrettably, knowledge of the limnology and hydrology of the lake (and others) (Beadle 1981a, b, Livingstone, Melack 1984, Coulter et al. 1986, Ribbink 1987); of nutrient budgets (Viner et al. 1981, Hecky 1984); of plankton (Hecky, Kling 1981, Lemoalle et al. 1981); of macrophytes (Gaudet et al. 1981); of the invertebrate fauna (Davies, Hart 1981) and of vertebrates (Leveque, Bruton 1981, Okemwa 1985, Coulter et al. 1986) is widely acknowledged as being inadequate and to have been outpaced by the development of the means to exploit the lakes and their resources. It is only the phytoplankton (Talling 1987) of Lake Victoria that have had comprehensive studies since 1898.

There is need for increased research to provide greater understanding that will minimize errors in managing the activities in the catchment that impact the water and its resources; to safeguard the resources and provide proper scientific basis for the planning and management of Lake Victoria aquatic resources. This will be a first step in trying to solve the 'problem' of the human activities impact on L. Victoria resources.

In this paper we try to highlight the human activities, their likely impacts and a need to find a solution to these problems.

THE AGRICULTURAL ACTIVITIES

One of the main pressing needs in developing countries is to increase food production to cater for an expanding population and the widespread undernourishment. This increase can be achieved in two ways: by intensification of agricultural activities and by involving new lands for agriculture. Intensified agriculture involves: applying irrigation techniques, increased and improved use of fertilizers, pesticides and other agrochemicals. Involvement of new lands includes land clearance and deforestation (Hamilton 1984), land reclamation from wetlands to create more fresh acreage for agriculture.

In Uganda where population growth has more than doubled within the last twenty years (from 7 to 15 m), it has been necessary to intensify agriculture. Large quantities of agrochemicals (pesticides) have been used to protect the cash and food crops, and others (fertilizers) to increase soil fertility and thus increase the yield of those crops (Bugenyi 1984a). The process of application and use of these agrochemicals has been mismanaged mostly by the local subsistence farmers with the consequent threat to the water environment.

Pesticide use (Bugenyi 1984a) leads to their unfavourable increase in the soils and eventually in waters and food chains. Tropical lakes are known to have high evaporation rates, in most cases accounting for more than 90% of the annual water loss (Coulter et al. 1986). This, therefore, increases the retention times of pollutants such as pesticides and this increases the susceptibility of the water body to the accumulation of pollutants. The rate of pesticide degradation ranges from weeks or months to several years.

Pesticide application to the crops in the acreage is generally inefficient in terms of the proportion of the dosage which actually reaches the target. The quantity and concentration used and the timing of treatment often applied as aerosols, smokes, sprays, paints, tree injections and granules have many chances of the pesticides missing the target and going to waste. These, then, reach into surface run-offs, streams and rivers and can be important sources of toxins. The behaviour of these is governed by sorption on soil particles, by dissolution in the water and by the equilibrium between sorption and solution. The pesticides in the waste gradually build up in the bodies of fish and other lacustrine organisms.

This problem is causing concern in several countries in bordering lakes in East Africa (Alabaster 1981). Though few quantitative studies have been done on body burdens, the matter has been considered in several symposia and workshops (Sserunjogi 1974, FAO/SIDA 1978, Meadows 1980, Dejoux et al. 1981). More quantitative estimates of pesticide toxin build-up in L. Victoria are urgently required because instances of pollution may be higher than realized, since there is very little monitoring of pesticide toxin levels. These particular toxins and others may cause, for instance, fish kills (Ochumba 1988), reduce fish reproductivity and elevate concentration of undesirable chemicals in the edible fish tissue. With little stock mobility among fish, localized toxin effects can be quite high especially in vicinities of urban industrialized centres.

There has been increased use of fertilizers in Uganda to try and increase crop yields. The chemical behaviour of nitrogen, N (in ammonium sulphate and calcium ammonium nitrate fertilizers) and phosphorus, P, is complex and dependent on a number of factors. Nitrogen in soils appears in three forms (organic, inorganic or ionic and gaseous) which interconvert.

The ionic forms (NO_3^+, NO_2^+ and NH_4^+) are water soluble and can be carried by the various land surface, subsurface and underground run-off to the rivers and lakes. Phosphorus compounds are much less soluble than those of N, and soil solutions contain small quantities of P. However, these relatively small quantitities of P and N can have an impact on the receiving waters.

Fertilizers, unlike other agrochemicals, have a rather more positive impact on plant growth in the receiving waters. The discharge of fertilizer wastes and the terrestrial run-offs from the farming catchments are known to cause explosive growth of algae and water weeds which is followed by periods of death and decay of the excessive biomass and the consequent deoxygenation of the already low quality water body. This, in essence, is what gives rise to "eutrophication" which is defined simply as "the enrichment of a water body with plant nutrients (mainly N and P) and the consequent deterioration in water quality due to prolific growth of aquatic plants". This phenomenon is certainly one of the most serious potential dangers to the L. Victoria waters. Some of the water quality parameters of the northern L. Victoria and L. Kyoga are displayed in Table 1. The increased biomass of plants temporarily arises from increased primary productivity. This is after trapping of the sun illumination by the green plants (water weeds, emergent vegetation, microscopic algae and papyrus) and through photosynthesis, organic compounds are produced. This organic food is utilized by the primary consumers (herbivorous invertebrates and fish) which are in turn fed on by the higher trophic fish communities. However, because of the much increased rate of supply of nutrients from the agricultural land, there is going to be excess biomass of the green plants (e.g. algae) and this is going to limit the sun illumination and a lot of dissolved oxygen is going to be used up when the algae begin to decay. Thus indications are that "eutrophication" is a potential threat to L. Victoria ecosystem (Bugeny, Mungoma 1980, unpubl.) in that with the consequent lack of enough dissolved oxygen there are likely to be frequent fish kills (Ochumba 1988) and a decline in fishery productivity. Further deterioration in water quality

Table 1. Mean physicochemical characteristics of Northern Lake Victoria and Lake Kyoga*

Temperature, ^{o}C	26	28
Turbidity, JTU	3.6	7.4
Secchi disk depth, cm	220	130
Dissolved O_2, mg/l	6	7.9
pH	7.4	7.3
Conductivity, µS/cm	105	122
Alkalinity, mg $CaCO_3$/l	55	72
Hardness, mg $CaCO_3$/l	30	38.5
Biological oxygen demand (BOD), mg O_2/l	1.2	-
$NO_3 \cdot N$, mg/l	1.2	3.10
$PO_4 \cdot P$, mg/l	0.2	0.41
$SO_4 \cdot S$, mg/l	5.0	32.5
SiO_2, mg/l	2.1	12.4

*Extracted from Mungoma (1988)

can be expected as a result of the decline of many herbivorous species over the years of heavy exploitation by fishing and predation.

The hazard of increased erosion of cultivated land in the catchment renders real the prospects of soil translocation to the lake (and other lakes; Coulter et al. 1986) whose euphotic zone will be changed and therefore its areal primary productivity and thus the essential fish yield.

The emerging scarcity of arable land due to increasing population density is creating intense pressure on the catchment areas. These areas have been cultivated year after year with the resultant top soil being vulnerable to wash-away by rain. The local farmers lack the know-how of soil erosion prevention. With these mismanaged agricultural practices on the land, there have been increased rates of erosion. There are other major sources of erosion (caused principally by road development and sometimes mechanical arable land treatment), quarry erosion (mining, although this is not yet a problem on a large scale in this region), sheet wash erosion (from other agricultural activities) and overgrazing. Forest (and vegetation) clearing for more arable land, wood for fuel, timber, building and other uses has led to vast lands to be exposed to soil erosion. The gradual destruction of Uganda's once extensive forests (Hamilton 1984) has turned some areas (e.g. Karamoja) from once well wooded lands into some of the world's most disastrous famine areas. There were many forested areas along the northern part of L. Victoria, e.g. the once extensive Mabira Forest Reserve. Through the above human activities, forest cover and wildlife habitats have been reduced. The resulting bare land has suffered soil erosion. There is, however, limited or actually no recent information regarding the absolute amount of eroded sediment into the L. Victoria water system. River and stream transport sediment or silt through one or two of these mechanisms: suspension, siltation (skipping or bouncing over the bottom) and bed load or gradual rolling and sliding of sediment along the river or stream bottom (Gottschalk 1964). Many major rivers and streams form flood plains from the seasonal rain falls that occur in the tropics. Turbulence associated with river flow usually ensures good mixing of water in the river channels. The annual pattern of flooding exerts a predominant effect on fish, influencing migration and growth (Balirwa, Bugenyi 1988). A major part of the silt load of rivers and streams is transported during floods. While this may have beneficial effects for agriculture initially, it certainly has adverse effects for fish. The fine silt in the water is dangerous to the fisheries in that it clogs the gills rendering them ineffective for respiration. Fish kills are common occurrences in such situations.

Lake Victoria is surrounded by extensive expanses of swamps or "adjacent wetlands". This zone is very vulnerable to human cultural or agricultural activities in the catchment areas. Impacts which may be significant when measured in terms of ecological and economic costs often prove difficult to detect in their early stages. This comes about as a result of failure to understand the underlying processes and linkages through which the cause-effect relationships manifest themselves. In most of these the wetland impacts can lead to irreversible

Table 2. Examples of catchment human activities

Human cultural activities	Physical result	Interim step (1)
Overgrazing in the catchment areas	Devegetation	Accelerated soil erosion
Urban development	Increased impervious surfaces	Increased erosion and water run-off
Dam construction	Water impoundment	Reduced flow
Irrigation development	1. Stream channelization	Increased flow
	2. Water diversion	Reduced flow
Industrial development	1. Thermal effluent	Increased ambient temperature
	2. Chemical pollution	Freshwater contamination
Agricultural development	Pesticide and fertilizer run-off	Freshwater contamination
		Freshwater enrichment
Other cultural activities (e.g. swamp burning for grazing, use of papyrus for paper and basketmaking), etc.	Wetland depletion, degradation and devegetation	Increased flow and eventual loss of filtration processes

resource degradation such as often occurs in cases of habitat destruction. Many adjacent wetland areas in the catchment of L. Victoria have been destroyed by burning and clearing for agricultural purposes and/or for other cultural purposes, e.g. getting papyrus reeds for house-thatching, basket and mats making, etc. This adjacent often narrow band of calm, well aerated and nutrient-rich water is of great ecological importance as it supports a large invertebrate fauna and is the feeding ground of many fish, especially the young and growing stages which also find protection from predators among submerged vegetation.

affecting the adjacent wetland environment

Interim step (2)	Principal effect and major significance
River-borne sediment transported	Increased sedimentation leading to eventual chocking and ecologically dead
Increased volume and velocity of river	Altered hydrological regime leading to reduced fish production
Reduced nutrient inputs	Reduced productivity, declining fishery production
Increased volume and velocity of river flow	Reduced productivity, plant species decline
Reduced nutrient inputs	
No step	Increased ambient temperature, death to flora and fauna
Contamination deposition and resuspension	Contaminated waters and later fish kills
Contamination deposition and resuspension	Contaminated waters, later fish kills and declining fish productivity
Potential algal overgrowth	Eventual eutrophication and gradual decline of fishery productivity
Flood-prone	Contaminated waters, loss of breeding areas for fish. Plant and animal species reduced

It should also maintain the good quality of water flowing through it (Talling 1957) to the adjacent open water, by regulating and filtering the inputs of pollutants from the catchment. Depending on the speed of flowthrough water, these wetlands have affected water quality (Gaudet 1978) and the effects have been shown to be dependent on season and rainfall, too. Thus, the agricultural activities talked about above are varied and many. These and many other cultural activities affecting the adjacent wetland environment first and later the open water body are given in Table 2 (Bugenyi 1987, unpubl.).

POTENTIAL WATER POLLUTION FROM INDUSTRIAL AND DOMESTIC WASTES

The Ugandan Lake Victoria drainage basin covers a large area (33,000 km^2), hence it becomes a potential source reservoir of pollutants to the water system especially through human activity. No water quality criteria have been determined for the water system, although measurements of certain parameters have been done at the Uganda Freshwater Fisheries Research Organisation (UFFRO), Jinja (Table 1); and at the national departments of fisheries and health. It is to be expected that certain bays and gulfs on the Western side of L. Victoria close to the areas of intense human activity (Table 3) are already experiencing pollution effects at different rates and levels.

The importation of pollutants into the lake is by the inflowing rivers, streams and rainfall surface run-offs. A survey of industrial activity which also includes mining and smelting around/within the lake basin (Alabaster 1981) should be undertaken. Information is further desired on parameters, the existing methods and levels of effluent treatment. In general, the organic residue originating from urban centres which have cropped up along the shores and within the drainage basin with populations of over 500,000 people is enormous. In some of these centres, e.g. Kampala city and Jinja town, there is partial treatment of domestic sewage effluent through biological oxidation ponds. In many of the other towns raw sewage empties directly into bays. In Kampala and Jinja sewage treatment plants are so situated that the partially treated sewage effluent is let through a stretch of papyrus swamp before it empties into the open waters. In this way, the swamp treats it further, especially when the flowthrough rate is slow, and yet some of these wetlands (swamps) are threatened by human activity.

Many of the industries let their mostly untreated effluents in the water, consisting of many organic compounds (most of which are biodegradable). Many of these industries are food processing (dairy and cooking oil industries and slaughterhouses or abattoirs, etc.), beer brewing and soft drinks manu-

Table 3. Main activities and likely types of pollution source in the western Lake Victoria basin

	Masaka	Kampala	Jinja	Other towns	Rural area
	commercial town	commercial and industrial city	mainly industrial town	mainly commercial towns	agricultural areas
Potential Pollution sources	Domestic sewage	Domestic sewage	Domestic sewage	Domestic sewage	Faecal pollution which results in bacterial, viral and parasitic diseases
	Coffee processing	Textiles	Textiles	Textiles	
	Others	Soap factories	Soap factories	Sisal	Pesticides:
		Oil mills	Oil mills	Beer and soft drinks	insecticides weedcides herbicides
		Breweries	Breweries	Dairy products	molluscides
		Dairy products	Paper and packaging	Others	Soil erosion resulting in siltation of waters
		Coffee	Steel rolling mills		
		Others	Starch and other flour mills		Irrigation-based diseases: bilharzia, malaria, onchocerciasis, etc.
			Tannery		
			Pharmaceuticals		Deforestation and overgrazing can result in soil erosion and hence siltation
			Others		
					Farms, e.g. sugar, coffee, tea, extensive farms which use pesticides and fertilizers

facture, sugar processing mills, fish meal plants and manure heaps. The effects of draining the effluents and urban run-offs from the areas around those industries are varied. They range from an increase in the biological oxygen demand (BOD) of the receiving waters which leads to low oxygen densities, high loads of the nutrient elements (N, P and S) and pathogens (from excreta) which are responsible for many waterborne diseases.

Jorgensen et al. (1982) have developed an environmental management model for the Upper Nile Lake system. The model was based on a great number of data obtained from Lakes Victoria, Kyoga and Albert. Among the sub-models is that of heavy metals. It particularly considers Cu from Kilembe Mines (Bugenyi 1979, 1982) and also the smelting plant in Jinja. The Lakes Edward-George system is connected to L. Albert by the R. Semliki.

The major part of heavy metal loading to the lakes occurs as suspended solids carried by rivers and streams. Similarly, the metals in the lakes are present mostly in particulate forms, either as sediments or as seston. But since the toxic effects of heavy metals (Cu and Cd; Bugenyi, Lutalo-Bosa 1987) are associated with the levels of ionic and other dissolved forms, the model and other experiments were focussed on the distribution between particulate and dissolved forms.

Cadmium has been associated with Cu as a by-product in Cu refining or smelting (GESAMP 1983). Emissions of Cd to the atmosphere have been through natural volcanic eruptions, forest fires and by burning of agricultural and municipal wastes including dried sewage sludge (Hutton 1982). And both Cu and Cd have been known to come from agricultural soils, mining waste and mine waters. There is, therefore, need to investigate and collect more data and make models for the movement and effects of heavy metals on the water biota, especially fish.

The general loading of allochthonous material (including pollutants noted above) and the budgets for water, plants and their nutrients are all indirectly affected by the geographical and hydrological nature of the lake. Relief is a factor that influences rainfall. In turn, rainfall influences the type of vegetation and this determines the type of soil in the region. To a large extent, too, it influences both the soil forming and soil destroying processes. This is part of the interactions between climatic and edaphic factors on one hand and biotic factors (e.g. vegetation) on the other.

Climatic features that have significant influences on the ecological processes in Lake Victoria basin should also be mentioned. There is a relationship between rainfall and evaporation. Given a suitable geomorphological situation, the extent and indeed the very existence of standing waters depend upon this relationship. The rate of evaporation depends on the temperature, vapour pressure, humidity of the atmosphere and wind stress at the surface. But the rate of evaporation per unit volume and the rate at which it may alter the composition of water, depend also on the relation between the surface area and volume of the lake. Over the years, Lake Victoria levels have changed and these had different ecological impacts on the lake (Welcomme 1969). There are annual lake level oscillations with maximum in May/June and minimum in October/November following the equinoctial rains. From 1961, the water level rose considerably. This rise was accompanied by short-term increases in catch per unit effort (CPUE) for the endemic Tilapia (Oreochromis esculentus, O. variabilis). It has also been shown that most species catches show two peaks within a year (Marten, Guluka 1975) and that there is a good correlation between catch time series, rainfall and lake level.

The annual cycle of stratification in temperate lakes is dominated by the great difference in radiant heat income between summer and winter. In Africa, wind-driven evaporation cooling generates much of the seasonal variation in the surface water temperature (Talling 1986, Beadle 1966), although influx of cool rain and reduced insolation during wet seasons can also be significant over a large part of Africa (Talling 1969). The specific thermal expansion of water is high at the normal temperatures of tropical lakes (Livingstone, Melack 1984); so a relatively small difference in temperature between the surface and deep water may provide considerable thermal stability and therefore stratification.

The range of water temperature in the tropical lakes is about 22° to 30°C (op. cit). Because of the diminishing capacity of water to retain O_2 as temperature increases, the absolute amount of O_2 per unit volume is less than in the temperate zone lakes. In addition, O_2 consumption is faster at higher temperatures. As a result Lake Victoria has a relatively smaller capacity for resisting O_2 demanding pollution (which, from above, happens to bear the main threats, e.g. input of high BOD demanding pollutants and those leading to eutrophication).

A sound basis needs to be laid for understanding the systematics, biography, hydrology and hydrobiology of a large part of lacustrine biota and from this we can hope to lay down plans for effective harvesting of the water resources and for the conservation of this environment. The means of harvesting the resources should be those that ensure this resource conservation.

FISHING ACTIVITIES AND THEIR EFFECTS ON RESOURCES

The earliest fishing was at subsistence level using basket traps, hooks and seine nets of papyrus with low efficiency carried out in swamps and river mouths. This type of fishing has now largely disappeared. At the beginning of the century gill-nets of cotton were introduced. Increases in population, rapid urbanization and improved means of communication increased the demand for fish and led to the rapid expansion of the fishing industry and to an increased fishing intensity. This development was specifically concentrated at Kavirondo Gulf (Kisumu), the northern coast and to Ssesse Islands, near to Kampala and Jinja. There was uncontrolled entry into the fisheries and this soon resulted in the decline in CPUE of accessible stocks (Ssentongo, Welcomme 1985).

The catches of Tilapia per net dropped from 30 fish in 1921 to six in 1928 in Kavirondo Gulf. Information on this development led to Graham's expedition of 1927-1929 (Graham 1929) which collected very valuable information during six months and confirmed the concentrated overfishing of Tilapia species. In 1933 the recommendation on mesh sizes came into force. From 1929 when Graham presented his proposals and up to the 1950s the catch of Tilapia per gill net had continued falling to 1.6

fish per net (in Kavirondo Gulf) because of increased fishing intensity and use of smaller meshed nets.

In 1952 synthetic fibre gill nets were introduced. These had a higher catching efficiency and a longer life span. Further, in 1953, motor boat engines were introduced of which the expected effect was a spreading of fishing activity. Beauchamp estimated in 1953 an increase in the fishing effort of 100% but only an increase in the yield of 10%.

The fisheries of Lake Victoria are mainly exploited by traditional and artisan fishermen. The fishermen have also engaged a destructive fishing method - beating of the water with a "tycoon" which drives all the fish into the net. Others have used the same fishing area all through. The overall effect has been a decline in catches, in sizes and probably in biomass.

The future of multispecies fishery in the lake is uncertain. It is very vulnerable to man-made changes. The complex biological and abiotic interactions underlying the richness of the faunas seem to be threatened by disregarding the pursuit of various socioeconomic gains. Fryer (1984) points out the way in which large-scale fisheries development schemes (which are envisaged in Uganda) have proceeded largely in ignorance of the biology and ecology of the fishes in these complex communities (Lowe-McConnell 1975).

Lake Victoria supports a complex multispecies fishery harvested with a variety of fishing gears and methods. Gill nets of varying mesh sizes, however, are most commonly used, although beach seines and cast nets are presently on the increase.

Some research work has been carried out on the fisheries of the lake. The results of exploratory bottom trawling in the lake indicated that water depth is an important variable affecting spatial distribution and catch rates. The catches of the endemic Tilapia (Oreochromis esculentus) and other tilapiine cichlids decline with increasing depth of 44.5 m beyond which the catches fall off again (Ssentongo, Welcomme 1985). The catch rates of one catfish, Bagrus docmac, follow a pattern similar to those of haplochromine species, whose biomass had been estimated by Kudhongania and Cordone (1974) and is now reduced by piscivory (Ribbink 1987). In subsequent studies of the northern part of Lake Victoria, Okaronon and Kamanyi (1988) observed that the catch rates of almost all the species in both commercial and experimental catches were on the decline.

Fisheries based on rivers, streams, wetlands and the littoral lake areas contribute significantly to fish landings in the region. The data on catches, however, is lacking. Among the fisheries which contribute to this are: Barbus species (C. 12 spp.), Labeo victorianus, Clarias mossambicus, Synodontis afro Fischeri and some smaller fishes such as the mormyrids. These are caught mainly at the river mouths and on the floodplain with a variety of fishing gear and methods (Whitehead 1958, van Someren 1962, Cadwalladr 1969). The lake shoreline and associated vegetation is a big source of the important Tilapia fishery, and at river mouths the fish caught include smaller clarids, Ctenopoma, Protopterus and several small fish species. All those species were abundant in the inshore areas during the 1940-1960s. Due to overfishing (and species introductions) catches of those species have drastically declined (EAFFRO Ann. Rep. 1957-1960, Garrod 1960, Marten 1979). Some work has been and is continuing to be done on the decline, the fishing methods employed, fish breeding, the biology and ecology of these species (Welcomme 1964, 1966; Cadwalladr 1965; Balirwa 1979, 1984). But, in the absence of a solid scientific platform it is extremely difficult to develop a sustainable fishery to meet the growing human requirements for protein. This has led to overfishing in the inshore areas and seems to have outpaced scientific management policy of the fisheries and other water resources.

And so, the present fishing patterns in the lake can be divided into a number of individual but overlapping species-oriented fisheries (Ssentongo, Welcomme 1985). Although experimental bottom trawling between the mid-1960s and mid-1970s had revealed a substantial quantity of Haplochromines, which could form a basis of a trawl fishery, there followed drastic declines in catches and stocks of the species in the 1980s (Witte 1981, 1983; Hoogerhoud et al. 1983, van Oijen 1982). These changes were attributed to several factors: (a) an intensive artisan fishery in the inshore water of less than 25 m depth around the lake, and this has been briefly explored; (b) introduction of exotic tilapiine cichlids since the 1950s and (c) the introduction of voracious Nile perch.

INTRODUCTION OF EXOTIC FISH SPECIES

The last four decades have witnessed numerous introductions of alien organisms into African freshwaters (Coulter et al. 1986). In some instances the effects of introductions of alien organisms are still unknown (and in a few the invaders appear to have increased fishery productivity). But the majority of introductions into natural systems have had unfortunate consequences or have posed serious problems of one kind or another (Barel et al. 1985, Ssentongo, Welcomme 1985, Coulter et al. 1986, Ribbink 1987). FAO/CIFA (1985) lists reasons for and dangers of introductions and provides a comprehensive summary of reported cases.

From about 1951, at least 5 non-indigenous Tilapias were introduced into Lake Victoria by the "Lake Victoria Fisheries Service Commission" and by the national Fishing departments, with the objective of improving the Tilapias. *O. leucostictus* caused least harm, while the others took over the habitats of and virtually replaced the endemic *O. variabilis* and *O. esculentus* (Benda 1979).

Nile perch, a voracious predator, was first introduced into Lake Kyoga in the mid-1950s. Despite opposition at the time (Jackson 1960, Fryer 1960), the species was introduced into Lake Victoria by the Uganda Department of Fisheries in about 1957 (Acere 1985). It is a voracious, opportunistic predator which grows to a very large size of almost 200 kg, length 1.9 m (Okemwa 1984). With abundance of prey, Nile perch populations have increased dramatically. Its introduction into L. Kyoga was to have been a test experiment to determine the effects that this predator would have on cichlid-dominated communities (Gee 1964, Stoneman, Rogers 1970, Acere 1985), but the introductions into L. Victoria took place before the test had run long enough for any form of evaluation to be made.

The diet of L. niloticus shortly after it was introduced into L. Kyoga was dominated by Haplochromine species, small mormyrids and the lungfish (Hamblyn 1966). By the mid to late 1960s, Oreochromis species formed the main food of L. niloticus (Gee 1964, 1969), and a little later Rastrineobola argentea (Okedi 1970, 1971). The effect of introductions are manifested in the catch composition (Table 4) compared to earlier studies of the early 1970s (Kudhongania and Cordone, 1974).

While Nile perch was largely responsible for the disappearance of the different fishes because of its opportunistic nature in preying upon a wide spectrum of other fish species (Ogari 1985), the contribution of overfishing and competition with other introduced tilapiines should not be ignored. Besides, the potential loss of vertebrate genetic diversity, as a result of this single ill-advised step is probably unparalleled in the history of man's manipulations of the ecosystem.

The biodynamic effects of the continuing disappearance of conventional species (e.g. Labeo victorianus, Oreochromis esculentus, etc.) and of the introduced species of L. niloticus and O. niloticus originate from a previous lack of management policy and the appearance of exotic species into the ecosystem (Kudhongania, Twongo 1985). Biological communities are structured along trophic hierarchies which influence every aspect of ecosystem function. Lake Victoria community and the water environment are in a period of transition which is being directed by the following current circumstances:

1. The influence of different management and development strategies as separately determined by the three riparian states.

2. Modifications imposed by the changing fishing regimes, including changes in gill net mesh sizes, uncontrolled entry and use of beach seines, illegal use of the "tycoon" in certain areas (and trawling). These changes arise from the differing needs for effective exploitation of specific categories of the stocks.

Table 4. Percentage composition of the four major fish species landed annually from Lake Victoria

Species	Country	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985
Lates	Kenya	0.5	1.1	4.5	14.0	16.0	54.4	54.4	67.7	57.6	56.5
	Tanzania	-	-	0.1	.	-	0.4	3.2	22.6	42.6	38.1
	Uganda	4.9	2.9	3.0	1.1	4.8*	0.9*	36.2	68.3*	52.8*	26.6*
Tilapines	Kenya	2.9	4.8	6.1	5.5	13.9	3.4	2.5	2.1	10.5	10.6
	Tanzania	7.0	8.2	13.7	12.5	16.5	18.8	16.6	9.3	9.0	10.8
	Uganda	16.7	19.1	19.9	9.9	9.1*	4.8*	11.9*	18.4*	36.5*	36.8*
Haplo-chromines	Kenya	34.1	32.4	27.8	21.6	13.5	4.2	4.2	0.8	0.06	0.01
	Tanzania	49.8	55,3	40.4	37.8	37.1	34.8	37.5	29.8	15.5	11.7
	Uganda	9.0	10.0	10.0	9.2	2.8*	3.1*	3.2*	0.3*	0.2*	-
Rastrineobola	Kenya	30.3	34.7	36.5	30.5	35.1	20.0	17.1	21.3	27.5	29.2
	Tanzania	0.5	3.8	3.4	2.8	2.3	5.3	3.5	4.5	1.6	8.0
	Uganda	0.1	0.2	-	0.4	0.2*	0.2*	2.2*	2.2*	6.5	34.6

*Records from landings at Masese near Jinja

3. The biodynamic effects of the continuing disappearance of conventional species and the introduction of the exotic species.

4. The impact of the inputs of allochthonous material from human activities along the lake shore and within the catchment area.

The cumulative effect of these and similar considerations has only partly started being investigated. There is urgent need to fully start comprehensive investigations so that a management policy for the lake can be drawn up.

CONCLUDING REMARKS

The main problem is that there has been a general decline of fish yields and virtual disappearance of certain species since the 1960s. This has been associated with overfishing, use of particular bad fishing methods, species introduction and land-based human activities. The extent to which these factors have affected the fisheries is unknown. In spite of this lack of organized information, the harvest or utilization of fisheries continues, with the introduced species tending to establish themselves with the reduced original species diversity in the ecosystem and the land-based activities increasing.

A comprehensive programme of research and data collection is urgently needed. The Canadian "International Development Research Centre" (IDRC) has funded a programme of research on "the bioecology of the introduced Nile perch". There is need to complement this programme. Lake Victoria being a multispecies fishery, there is need to have an understanding of the gears in operation and their effects on the lake multispecies fisheries. The concrete causes of the disappearance of species (e.g. *O. esculentus*, *L. victorianus*) and the decline of others, and the possibility of re-introduction of the formerly endemic species as a means of reviving them should be investigated.

With changes in percentage of species composition involving the haplochromines, *L. niloticus*, the Tilapiines, and *Rastrineobola argentea* as outlined above, there is need to intensively study and determine the possible causes of these changes. The complexity of L. Victoria fisheries calls for constant monitoring of the trends including changes in fishing inputs.

The close ecological relationships between lotic, land/water ecotones or interfaces on the one hand, and the open water on the other require a balanced study of related aspects. Apart from the few studies briefly cited above, there is lack of quantitative information on the fishery resources of the riverine and wetlands (land-water ecotones). There is need for resource evaluation combined with an examination of the level of harvest of fisheries. Investigation of the breeding biology and ecology of the commercially more important species which are on the decline, as well as the introduced species in their host environment should be undertaken.

To complement the above studies, there is need to study the baseline chemical-physical conditions of, and the long-term impact of seasonal changes on discharge, nutrient concentrations, pH, temperature and dissolved substances on the composition and abundance of biota in the water on a temporal and spatial basis.

A data bank for the distribution in the water of the flora and fauna, and of sediment composition should be established and monitored for such pollutants as pesticides, heavy metals and other organic and inorganic chemical pollutants. Toxicological studies on the fisheries should be carried out for some of those. Their cycling in the terrestrial/aquatic systems should be ascertained.

The introduced fish species have contributed to changes in the trophic dynamics of the Lake Victoria ecosystem. It is not known how much for instance the introduced species have affected primary production and vice versa. A thorough study of water quality characteristics, primary production and invertebrates would solve the problem especially as there are some old data with which to compare the present situation.

To augment the IDRC programme, it is necessary to examine factors affecting the growth and abundance of zooplankton and benthos which are fed on by Nile perch and other fishes.

The above data would provide the validation and calibration of the Lake Victoria model developed but never calibrated by

Jorgensen et al. (1982). This would consequently enable an evaluation of the trophic and pollution state of the lake against data of the 1950s and 1970s. Management options will emerge from all these investigations.

ACKNOWLEDGMENTS

This paper is a result of a survey of specialized works in various disciplines of hydrobiology and fisheries. We wish to express our gratitude to those who generously contributed ideas through discussion of the problem, especially colleagues at UFFRO and in the Fisheries Department, those from other sectors such as the Ministry of Environment, Makerere University and the National Water and Sewerage Cooperation. We express thanks to Mrs. F. Balirwa for typing through various drafts of the manuscript.

REFERENCES

Acere, T.O. (1985): Observation on the biology, age, growth, maturity and sexuality of Nile perch Lates niloticus (Linné), and the growth of its fishery in the northern waters of Lake Victoria. In: CIFA report of the Third Session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria, Jinja, Uganda, 4-5 October, 1984, pp. 46-61. Fish. Rep. 335. FAO, Rome.

Alabaster, J.S. (1981): Review of the state of aquatic pollution of East African inland waters. CIFA Occas. Pap. 9, pp. 1-36.

Anderson, A.M. (1961): Further observations concerning the proposed introduction of Nile perch into Lake Victoria. E. Afr. Agric. For. J. 26, 195-201.

Balirwa, J.S. (1979): The food of six Cyprinid fishes in Lake Victoria. Hydrobiologia, 66, 65-72.

Balirwa, J.S. (1984): Ecological separation in Barbus species in Lake Victoria. M. Sc. Thesis, University of Dar es Salaam, 155 pp.

Balirwa, J.S., Bugenyi, F.W.B. (1988): An attempt to relate environmental factors to fish ecology in the lotic habitats of Lake Victoria. Verh. Internat. Verein. Limnol. 23.

Barel, C.D.N., Dorit, R., Greenwood, P.H., Greenwood, G., Fryer, G., Hughes, N., Jackson, P.B.N., Kawanabe, H., Lowe-McConnell, R.H., Nagoshi, M., Ribbink, A.J., Trewavas, E., Witte, F., Yamaoka, K. (1985): Destruction of fisheries in Africa's lakes. Nature 315, 19-20.

Beadle, L.C. (1966): Prolonged stratification and de-oxygenation in tropical lakes. Crater Lake Nkugute, Uganda, compared with Lakes Bunyonyi and Edward. Limnol. Oceanogr. 11, 152-163.

Beadle, L.C. (1981a): The inland waters of tropical Africa. 2nd edition. Longman, New York, 475 pp.

Beadle, L.C. (1981b): Limnology of African inland waters. In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds): The ecology and utilization of African inland waters, pp. 11-16. UNEP Reports and Proceedings, Series 1, Nairobi.

Benda, R.S. (1979): Analysis of catch data from 1968 to 1976 from nine fish landings in the Kenya waters of Lake Victoria. J. Fish. Biol. 15, 385-387.

Beauchamp, R.S.A. (1953): Annual Report (1953) of East African Freshwater Fisheries Research Organisation.

Bugenyi, F.W.B. (1979): Copper ion distribution in the surface waters of Lakes George and Idi Amin. Hydrobiologia, 64, 9-15.

Bugenyi, F.W.B. (1982): Copper pollution studies in Lakes George and Edward, Uganda: The distribution of Cu, Cd and Fe in the water and sediments. Environ. Pollut. (Ser. B) 3, 129-138.

Bugenyi, F.W.B. (1984a): Potential aquatic environmental hazards of agro-based chemicals and practices in a developing country, Uganda. Proc. Int. Conf. Environm. Haz. Agrochem. Vol. 1, pp. 461-73.

Bugenyi, F.W.B. (1984b): Copper distribution and pollution studies in Lakes George and Edward, Uganda. Ph.D. Thesis. Makerere University, Kampala, 132 pp.

Bugenyi, F.W.B. (1987): Human activities and the detrimental impacts inflicted on the Nile basin and its resources in Uganda. Unpublished MS.

Bugenyi, F.W.B., Mungoma, S. (1980): Preliminary general limnological investigations of the freshwaters adjacent to Jinja industrial town. Unpublished MS.

Bugenyi, F.W.B., Lutalo-Bosa, A.J. (1987): Cu, Cd and Fe in aquatic systems of Uganda, East Africa (in press - SCOPE Proceedings of Heavy Metals Workshop, New Delhi, India, 16-20 February 1987).

Cadwalladr, D.A. (1965): Notes on the breeding biology of Labeo victorianus Boulenger (Pisces: Cyprinidae) fishery of Lake Victoria. Rev. Zool. Bot. Afr. 72 (1-2), 109-34.

Cadwalladr, D.A. (1969): A discussion of possible management methods to revive the Labeo victorianus fishery of Lake Victoria, with special reference to the Nzoia River, Kenya. Occas. Pap. Fish. Dep. Uganda, 2, 1-4.

Coulter, G.W., Allanson, B.R., Bruton, M.N., Greenwood, P.H., Hart, R.C., Jackson, P.B.N., Ribbink, A.J. (1986): Unique qualities and special problems of the African Great Lakes. Env. Biol. Fish. 17, 161-84.

Davies, B.R., Hart, R.C. (1981): The ecology of aquatic invertebrates. In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds):

The ecology and utilization of African inland waters, pp. 51-68. UNEP Reports and Proceedings, Series 1, Nairobi.

Dejoux, C., Deelstra, H., Wilkinson, R.C. (1981): Pollution. In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds): The ecology and utilization of African inland waters, pp. 147-161. UNEP Reports and Proceedings, Series 1, Nairobi.

EAFFRO (1957-1960): Annual Reports of East African Freshwater Fisheries Research Organisation.

FAO/CIFA (1985): Introduction of species and conservation of genetic resources. Committee for Inland Fisheries of Africa, Sixth session, Lusaka, Zambia, 7-11 October, 1985, CIFA/85/13, FAO, Rome, 18 pp.

FAO/SIDA (1978): Proceedings: 6th FAO/SIDA Workshop on Aquatic Pollution in Relation to Protection of Living Resources. Nairobi and Mombasa, Kenya. FAO FIR: TPLR 78, Rome.

Fryer, G. (1960): Concerning the proposed introduction of Nile perch into Lake Victoria. E. Afr. Agric. J. 25, 267-70.

Fryer, G. (1984): The conservation and rational exploitation of the biota of Africa's great lakes. In: A.V. Hall (ed.): Conservation of threatened natural habitats, pp. 135-154. South Africa National Scientific Programmes Report 92. CSTR, Pretoria.

Garrod, D.J. (1960): The fisheries of Lake Victoria, 1954-59. E. Afr. Agric. For. J., 26, 142-8.

Gaudet, J.J. (1978): Effects of a tropical swamp on water quality. Verh. Internat. Verein. Limnol. 20, 2202-6.

Gaudet, J.J., Mitchell, D.S., Denny, P. (1981): Macrophytes (aquatic vegetation). In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds): The ecology and utilization of African inland waters, pp. 27-36. UNEP Reports and Proceedings. Series 1, Nairobi.

Gee, J.M. (1964): Nile perch investigation. Ann. Rep. E. Afr. Freshwat. Fish. Res. Org., 1964, 13-7.

Gee, J.M. (1969): A comparison of certain aspects of the biology of Lates niloticus (Linné) in some East African Lakes. Rev. Zool. Bot. Afr., 80, 244-62.

GESAMP (IMO/FAO/UNESCO/WHO/IAEA/UN/UNEP) (1983): Review of potentially harmful substances, cadmium, lead and tin. Reports and studies 22.

Gottschalk, L.C. (1964): Reservoir sedimentation. In: Chow, V.T. (ed.): Handbook of Applied Hydrology. McGraw-Hill, New York, pp. 17.2-17.34.

Graham, M. (1929): The Victoria Nyanza and its fisheries 1927-28. London, Crown Agents for the Colonies.

Hamblyn, E.L. (1966): The food and feeding habits on Nile perch, Lates niloticus (Linné) (Pisces: Centropomidae). Rev. Zool. Bot. Aft., 74 (1-2), 1-28.

Hamilton, A.C. (1984): Deforestation in Uganda. Oxford University Press, 95 pp.

Hecky, R.E. (1984): African lakes and their trophic efficiencies: a temporal perspective. In: Meyers, D.G., Strickler, J.R. (eds): Trophic interactions within Aquatic ecosystems, pp. 405-448. AAAS selected symposium 85, Westview Press, Denver.

Hecky, R.E., Kling, H.J. (1981): The phytoplankton and protozooplankton of the euphotic zone of Lake Tanganyika, Species composition, biomass, chlorophyll content, and spatiotemporal distribution. Limnol. Oceanogr., 26, 548-64.

Hoogerhoud, R.J.C., Witte, F., Barel, C.D.C. (1983): The ecological differentiation of two closely resembling Haplochromis species from Lake Victoria (H. iris and H. hiatus; Pisces, Cichlidae). Neth. J. Zool. 33, 283-305.

Hutton, M. (1982): Cadmium in the European Community, London, Chelsea College, Monitoring and Assessment Research Centre, 99 pp. (MARC Report No. 26).

Jackson, P.B.N. (1960): On the desirability or otherwise of introducing fishes to waters that are foreign to them. CSA Symposium Hydrobiol. Inl. Fish. CSA/CCTA Publ. 63, pp. 157-64.

Jorgensen, S.E., Kamp-Nielsen, L., Jorgensen, L.A., Mejer, H.F. (1982): An environment management model of the Upper Nile Lake System. ISEM Journal 4 (3-4), 5-72.

Kudhongania, A.W., Cordone, A.J. (1974): Batho-spatial distribution patterns and biomass estimate of the major dermasal fishes in Lake Victoria. Afr. J. Trop. Hydrobiol. Fish. 3, 15-31.

Kudhongania, A.W., Twongo, T.K. (1985): Some considerations for research and management of the Lake Victoria fisheries. In: FAO/CIFA report of the third session of the sub-committee for the development and management of the fisheries of Lake Victoria, Jinja, Uganda, 4-5 October 1984, pp. 139-142. Fish. Rep. 335, FAO, Rome.

Lemoalle, J., Adeniji, A., Compere, P., Ganf, G.G., Melack, J., Talling, J.F. (1981): Phytoplankton. In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds): The Ecology and Utilization of African Inland Waters, pp. 37-50. UNEP Reports and Proceedings, Series 1, Nairobi.

Leveque, C., Bruton, M.N. (1981): Fishes. In: Symoens, J.J., Burgis, M., Gaudet J.J. (eds): The Ecology and Utilization of African Inland Waters. UNEP Reports and Proceedings, Series, 1, Nairobi.

Livingstone, D.A., Melack, J.M. (1984): Lakes of subsaharan Africa. In: Taub, F.B. (ed.): Ecosystems of the World: Lakes and Reservoirs, pp. 467-497. Elsevier, Amsterdam.

Lowe-McConnel, R.H., (1975): Fish communities in tropical freshwater. Longman, London, 337 pp.

Marten, G.G. (1979): Impact of fishing on the inshore fishery of Lake Victoria (East Africa). J. Fish. Res. Board Can. 36, 891-900.

Marten, G.G., Guluka, L. (1975): Fluctuations in fish catches and prices and their correlations with climatic factors. Ann. Rep. E. Afr. Freshwat. Fish. Res. Org., 1974, 69-75.

Meadows, B.S. (1980): The quality of water in the Lake Victoria Basin and its protection. Seminar ser. Lake Victoria Basin Dev., Inst. Dev. Stud. Univ. Nairobi, 7. Rev. 1, 1-16.

Mungoma, S. (1988): Horizontal differentiation in the limnology of a tropical river lake (Kyoga, Uganda). Hydrobiologia, 162, 89-96.

Ochumba, P.B.O. (1988): Observations on the Kenyan part of Lake Victoria during massive fish deaths between 1984-87. Abstracts p. 123. 3rd International Conference on the Conservation and Management of Lakes, "Balaton" '88.

Ogari, J. (1985): Distribution, food and feeding habits of Lates niloticus in Nyanza Gulf of Lake Victoria (Kenya). In: FAO/CIFA Report of the Third session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria. Jinja, Uganda, 4-5 October, 1984, pp. 68-80. Fish. Rep. 335, FAO, Rome.

Ogutu-Ohwayo, R. (1985): The effects of predation by Nile perch, Lates niloticus (Linné) introduced into Lake Kyoga and Lake Victoria. In: FAO/CIFA Report of the Third Session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria. Jinja, Uganda, 4-5 October, pp. 18-41. 1984. Fish. Rep. 335, FAO, Rome.

Oijen, M.J.P. van (1982): Ecological differentiation among the piscovorous haplochromine cichlids of Lake Victoria (East Africa). Neth. J. Zool. 32, 335-363.

Okaronon, J.O., Kamanyi, J. (1986): Recent trends in the fisheries of the northern portion of L. Victoria (Uganda). UFFRO Seminar, November, 1986.

Okedi, J. (1970): Further observations on the ecology of Nile perch (Lates niloticus, Linné) in Lakes Victoria and Kyoga. EAFFRO. Annual report 1970, pp. 42-55.

Okedi, J. (1971): Further observations on the ecology of the Nile perch in Lake Victoria and Lake Kyoga. E. Afric. For. J., 130, 42-45.

Okemwa, E. (1984): Potential fishery of Nile perch Lates niloticus Linné (Pisces: Centropomidae) in Nyanza Gulf of Lake Victoria, East Africa. Hydrobiologia, 108, 121-126.

Okemwa, E. (1985): Principal gaps in the knowledge of fisheries of the Kenyan waters of Lake Victoria. In: FAO/CIFA Report of the Third Session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria, Jinja, Uganda, 4-5 October, 1984, pp. 177-222, Fish. Rep. 335, FAO, Rome.

Ribbink, A.J. (1987): African lakes and their fishes: conservation scenarios and suggestions. Env. Biol. Fish., 19, 3-26.

Someren, V.D. van (1960): Nile perch studies. Ann. Rep. E. Afr. Freshwat. Fish. Res. Org., 1960, 7-8.

Ssentongo, G.W., Welcomme, R.L. (1985): Past history and current trends in the fisheries of Lake Victoria. In: FAO/CIFA Report of the Third Session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria. Jinja, Uganda, 4-5 October, 1984, pp. 123-138. Fish. Rep. 335, FAO, Rome.

Sserunjogi, J.M.S. (1974): A study of organochlorine insecticide residues in Uganda with special reference to Dieldrin and DDT. Proc. Symp. Nucl. Technol. in Comp. Food and Env. Cont. FAO/IAEA/WHO Finland, IAEA, STI/PUB348, pp. 43-48.

Stoneman, J. Rogers, J.F. (1970): Increase in fish production achieved by stocking exotic species (Lake Kyoga, Uganda). Occas. Pap. Fish. Dep. Uganda (3), 16-9.

Talling, J.F. (1957): Some observations of stratification of Lake Victoria. Limnol. Oceanogr., 3, 213-21.

Talling, J.F. (1966): The annual cycle of stratification and phytoplankton growth in Lake Victoria (East Africa). Int. Rev. Hydrobiol., 51, 545-621.

Talling, J.F. (1969): The incidence of vertical mixing, and some biological and chemical consequences, in tropical African lakes. Verh. Int. Ver. Limnol. 17, 998-1072.

Talling, J.F. (1987): The phytoplankton of Lake Victoria (East Africa). Arch. Hydrobiol. Beih. Ergebn. Limnol., 25, 229-256.

Viner, A.B., Breen, C., Golterman, H.L., Thornton, J.A. (1981): Nutrient budgets. In: Symoens, J.J., Burgis, M., Gaudet, J.J. (eds): The Ecology and Utilization of African Inland Waters, pp. 137-48. UNEP Reports and Proceedings, Series, 1, Nairobi.

Welcomme, R.L. (1964): Notes on the present distribution and habits of the non-endemic species of Tilapia which have been introduced into Lake Victoria. E. Afr. Freshwat. Res. Org. Ann. Rep., 1962, 63.

Welcomme, R.L. (1966): The biological and ecological effects of climatic changes on some fishes in the Lake Victoria basin. Ph. D. Thesis, University of East Africa.

Welcomme, R.L. (1969): The effect of rapidly changing water level in Lake Victoria upon the commercial catches of Tilapia (Pisces: Cichlidae). In: Obeng, L. (ed.): Accra Symposium. Accra Ghana Universities Press for Ghana Academy of Science, pp. 242-250.

Witte, F. (1981): Initial results of the ecological survey of the haplochromine cichlid fishes from the Mwanza Gulf of Lake Victoria (Tanzania): breeding patterns, trophic and species distributions. Neth. J. Zool. 31, 175-202.

Witte, F. (1984): Ecological differentiation in Lake Victoria haplochromines: a contribution to the comparison of cichlid species flocks in African Lakes. In: Echelle, A.A., Kornfield, I. (eds): Evaluation of Fish Species Flocks. University of Maine at Orono Press, Orono.

Witte, F., Goudswaard, P. (1985): Aspects of the haplochromine fishing in southern Lake Victoria. In: FAO/CIFA report of the Third Session of the Sub-committee for the Development and Management of the Fisheries of Lake Victoria. Jinja, Uganda, 4-5 October, 1984, pp. 81-88. Fish. Rep. 335, FAO, Rome.

Whitehead, P.J.P. (1958): The indigenous river fishing methods in Kenya. E. Afr. Agric. For. J. 24, 111-120.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

PROTECTION AND UTILIZATION OF THE JIUZHAIGOU MOUNTAIN LAKE GROUP

DU HENGCHAN, LIU LIU,* LI LONGYAAN**

Sichuan Provincial Construction Committee; *Sichuan Learned Society of Environmental Science, 31 Xiyujie, Changdu; **Sichuan Research and Monitoring Institute of Environmental Protection, 18 Section 4, Renmin Nanlu, Chengdu, China

INTRODUCTION

Jiuzhaigou (Nine Village Valley) is situated in the southern part of Minshan Mountain Chain, Nanping County, Sichuan Province, about 400 kilometres from Chengdu (Fig. 1). The Jiuzhaigou River is a headwater of the Baishuihe River (a tributary of the Jialingjiang River). The main stream is 45 km long with a catchment area of 650 km^2 and an annual mean flow of 9 m/s. The Jiuzhaigou is a newly-developed scenic spot characterized by beautiful mountain barrier lakes. It was listed as a national scenic spot in 1982.

The Jiuzhaigou is surrounded by mountains. The altitude of the peaks inside the territory ranges from 3500 to 4746 m a.s.l. The area is draped by forest up to 42.6%. The main stream is Y-shaped (Fig. 2), with water running from south to north. Hundred and three various lakes are concentrated in the valley. The main lakes are Long Lake, Mirror Lake, Rhinoceros Lake, Sparkling Lake, Shuzhengqun Lake, Five-flower Lake, Multi-colored Lake, Swan Lake and Reed Lake. Among them Long Lake is not only the highest (3103 m) but also the largest one. It is 4750 m long and of an area of 99.9 ha. There are a number of waterfalls (cascades), rapid flows, travertine rapids, creeks and grassy shallows between the lakes. Of all the waterfalls, the Nuorilang, Pearl Rapids, Panda Lake, Shuzheng and Shuzhengqun Lake waterfalls are the most famous. They cut through groves and rush down. The area is a combination of an infinite variety of fantastic landscapes, with a breathtaking

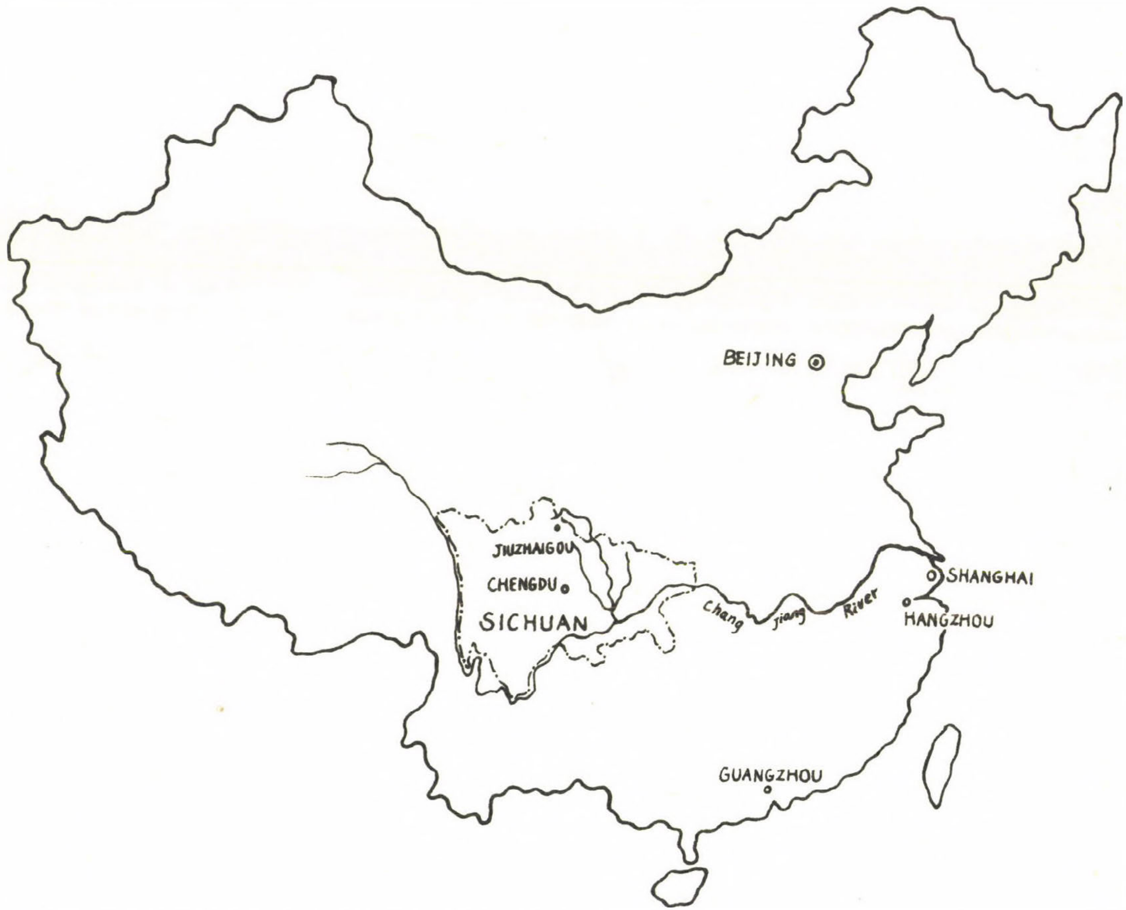


Fig. 1. Geographic location of Jiuzhaigou

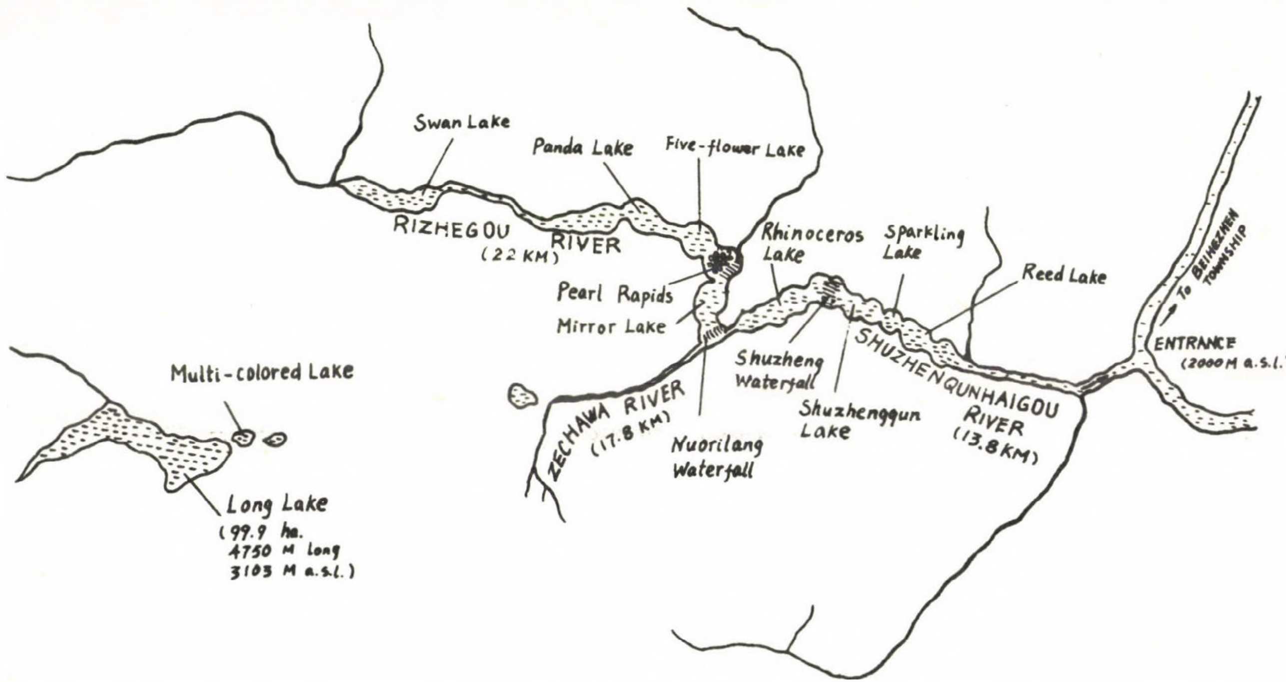


Fig. 2. Jiuzhaigou (Nine Village Valley)

scenery. Pearl Rapids is the largest travertine rapid. Reed grows well at the slow-flowing and shallow parts of the water. The water of the Jiuzhaigou lakes is of a low degree of mineralization (of the bicarbonate type). It is very clear and turquoise blue. Water quality is basically of Class 1, National Standard for Surface Water (Table 1). Air quality is good, coming up to Class 1, National Standard for Ambient Air (Table 2).

Table 1. Water quality of the Jiuzhaigou lake group

Item	x_{min}	x_{max}	Unit
pH	7.59	8.25	pH
SS	0.50	8.30	mg/l
DO	86.6	110.7	%
NO_3^-	0.035	0.290	mg/l
NO_2^-	<0.005		mg/l
NH_3	0.0031	0.015	mg/l
SO_4^{2-}	11.2	31.8	mg/l
HCO_3^-	145	312	mg/l
N(K)	0.120	0.27	mg/l
F	0.227	0.417	mg/l
Mg	3.71	18.4	mg/l
P	0.0125	0.079	mg/l
K	0.35	0.92	mg/l
Ca	36.1	68.1	mg/l
Cr	0.18	8.30	µg/l
Mn	0.84	14.99	µg/l
Fe	0.01	0.22	mg/l
Ni	0.25	14.30	µg/l
Cu	0.36	3.72	µg/l
Zn	0.01	0.17	mg/l
As	<0.05		µg/l
Cd	0.006	0.057	µg/l
Hg	<0.001		µg/l
Pb	0.005	2.19	µg/l

*Data in Tables 1 and 2 are from a report written by a special group led by Mr. Liao Ji, Sichuan Research and Monitoring Institute of Environmental Protection.

Table 2. Air quality of the Jiuzhaigou

Item	X_{min}	X_{max}	Unit
TSP	0.03	0.08	mg/nm^3
SO_2	0.003	0.008	mg/nm^3
NO_x	0.0026	0.0069	mg/nm^3
CO	4.05		mg/nm^3
Photochemical oxidant	0.0031	0.0090	mg/nm^3

The valley is inhabited by 800 or more Tibetans. It is very interesting to see their distinctive and simple life style. The blending of picturesque natural scenery and primitive life style makes it a magnificent touristic site. Once a guide remarked that of all the beautiful scenic spots the world has ever offered, such an attractive sight as the Jiuzhaigou can scarely be found. It is reputed to be a Fairyland in China.

ORIGIN OF JIUZHAIGOU

The Jiuzhaigou is of limestone geology. The peculiar terrain and the number of mountain cascade lakes formed as a result of the interactions of Himalayan orogenesis, glaciation, earthquake, debris flow and weathering.

Water is the chief landscape of the Jiuzhaigou scenic spot. Abundance and good quality of water make the water body into a bright and colourful landscape of mountain lakes. The immense quantity of water originates from its thick forest, from the plentiful precipitation (annual mean: 780 mm) and meltwater. In the forest humus soil and mosses pile up into a layer of several tens of centimetres. Rainwater and meltwater fall down to the ground and infiltrate these surface layers. Spring and creek water of good quality are flowing through, then passing into the lakes.

FACTORS IMPACTING THE JIUZHAIGOU MOUNTAIN LAKE GROUP

Good quality water is the peculiarity of the Jiuzhaigou mountain lake group. Therefore, maintaining water quality and quantity is the central task of protecting the lakes. There are three main factors impacting water quantity and quality.

1. Forest Devastation. The Jiuzhaigou had been a felling area before it was opened as a touristic site. Part of the trees was felled. Local residents have not so far stopped cutting down some trees for fuel. Besides, potential forest fire is also a threat to the territory. Forest is one of the origins of water. If it is devastated, the deterioration of water body is inevitable.

2. Debris Flow. Debris flow in the Jiuzhaigou is caused both by natural and artificial factors. Natural debris flow is the consequence of the interactions of multiple factors, such as frost, weathering, earthquake and rainfall. Artificial debris flow is the consequence of forest devastation and slope vegetation destruction. Debris flow is extremely harmful to lakes. It pollutes water sources, fills up lakes, devastates forests and damages highways and all sorts of equipment. There are 27 kinds of debris flow of medium or small scale. They have changed some lakes into meadows. Reed Lake is becoming a small aged lake. The grassy rapids of Mirror Lake and Rhinoceros Lake extend year by year. All these are caused by debris flow.

3. Tourist Pollution. Since it was opened for the public, Jiuzhaigou has become a famous scenic spot for tourism, being well known both at home and abroad. The Jiuzhaigou Scenic Territory and the Yosemite National Park of the USA established the relation of a sister park in May, 1988. It is estimated to attract over a hundred thousand tourists from home and abroad in 1988. Due to the rapid increase of tourism, solid wastes, wastewater and waste gases produced by human activities will have an impact on its environment, particularly the water body. In addition, inadequate buildings and constructions will also impair the natural scenery.

MEASURES TO PROTECT THE JIUZHAIGOU MOUNTAIN LAKE GROUP

A series of comprehensive measures have been adopted by the Chinese government to protect the Jiuzhaigou, especially its valuable mountain lake group. First, the area was approved to be a national scenic spot. The Jiuzhaigou Natural Conservation and Administration Bureau was founded immediately. It adopted a series of strict regulations, drew up the overall plan of protecting, exploiting and constructing the Jiuzhaigou, and appointed 80 technicians and administrators. For protection of the water body, the following measures have been taken.

1. Forest Conservancy. Timber felling has been prohibited inside the valley since 1979. The bureau has devoted much attention to recovering forest vegetation. It has begun to reafforest in the felled area. For five years, about 25,000 ha have been reafforested. The chief species are fir and spruce. The young trees are growing well. In the Jiuzhaigou, the weather is dry in winter and spring. During this period trees easily catch fire. So the period from November to next May is declared to be a fire-endangered period. Teams consisting of professional fire fighters and local residents are responsible for fire protection. As a result, no forest fire has occurred for five years. The Sichuan Provincial Government allocated funds to pave a new highway leading to a brushland outside the valley to solve the problem of the local people to be able to cut trees for fuel. Felling in the scenic territory is strictly prohibited.

2. Debris Flow Control. The Sichuan Provincial Government has devoted much attention to the problem of debris flow in the Jiuzhaigou and allocated 2.74 million yuan (RMB) to control it. The Chengdu Institute of Mountain Disaster and Environment, Academia Sinica, is responsible for the study and control of debris flow in the Jiuzhaigou. There are 14 kinds of debris flow requiring supervision, of which 7 are kept under control. Measures combined with civil engineering and biology (construction of dams and planting of trees on the slopes) have been adopted, and they have achieved initial success.

3. Tourist Pollution Control. For reducing pollution made by tourists, the principal "Tour in The Valley while Stay out

of it" was organized when working out the overall programme of the Jiuzhaigou scenic spot. This implies that it is not permitted to build any new hotels in the valley. Board and lodging facilities, such as hotels and restaurants, are built in the Beihezhen township outside the valley. During the day tourists go sightseeing in the valley, then come back to hotels outside the valley in the evening. A number of paths, pavilions, toilets and garbage bins have been set up, and some persons are specially assigned for managing them.

4. To preserve the primitive beauty of the Jiuzhaigou scenery as well as its environment, artificial landscapes are not built as much as possible. Some buildings which may impair the scenery or pollute the environment have been dismantled. The few necessary facilities for sightseeing must be harmonious with the natural environment concerning size, shape, features, colour and material. After having been protected, exploited and managed for only five years, a tourist system has been established in the Jiuzhaigou. Now it is capable of receiving a lot of tourists. Although about three hundred thousand tourists both from home and abroad have visited the territory, the environment and scenery are well protected. Forests are much thicker than before, the water body of the lakes has never been polluted.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

STATE OF THE ECOSYSTEM OF LAKE BAIKAL AND ITS CATCHMENT AREA. PROBLEMS OF CONSERVATION AND RATIONAL USE OF RESOURCES

G.I. GALAZY

Ecological Museum, Siberian Branch of the Academy
of Sciences of the USSR
Irkutsk Region, Listvyanka 666016, USSR

In the Baikal region industrial development should undergo in harmony with the conservation of this unique ecosystem.

Lake Baikal is the largest body of the purest freshwater on the Earth. Its volume is 23,000 km^3, representing one-fifth of the global resources of surface freshwater and more than 80% of that in the Soviet Union.

Water residence time in Lake Baikal amounts to 400 years. This means that all of the pollutants contaminating the lake remain there for many centuries, and if changes take place in the ecosystem they are irreversible.

Up to the present time more than 2,500 animal and plant species have been identified in Lake Baikal. Three-fourths of them are endogenous.

The purity of the lake water is maintained by the biological activity of the organisms which have adapted throughout millions of years to the severe environmental conditions, i.e. low surface temperature (average and maximum annual temperature being about 4 and 14-15 $^{\circ}C$, respectively), low temperature in the deepwater areas (less than 3.6 $^{\circ}C$ in depths more than 250-300 m), high transparency (the maximum being 40 m), slow mineralization and stable chemical composition of the inflowing waters. At the same time, the continuous pollution of Lake Baikal and its basin has been alarming for more than 20 years.

The main sources of pollution are as follows:

- Industrial waste water discharge of the pulp and paper mill at Baikalsk (about 420,000 m^3 day^{-1}).

- Industrial waste water discharge of the Ulan-Ude region (over 120,000 m^3 day^{-1}).

- Industrial waste water discharge of the pulp and paper mill at Selenga (about 60,000 m^3 day^{-1}).

- Erosion induced by surface water runoff and wind, wash-out of soils and toxic substances.

- Timber rafting on the lake, erection of rafts at the mouths of tributaries.

- Continuously increasing atmospheric pollution, about 6% of which is of industrial origin at present.

- Pollution associated with the construction of the Baikal-Amur Railway.

- Pollution associated with the large fleet on the lake.

- Deterioration caused by the development of uncontrolled tourism.

A number of government decrees has been adopted aiming at a more rational utilization as well as conservation of natural resources.

Realization of this program has already been started by the following measures:

- Rafting of wood across Lake Baikal and its tributaries has been stopped. River beds have been cleaned of sunken wood, resulting in an improvement of the water quality of the tributaries. Due to this improvement, some commercial fish species (e.g. omul, grayling) spawn again in the rivers Barguzin, Goloustnaya, Itanez. New rules of wood exploitation have been worked out and adopted in the Baikal region.

- Losses of bark and wood were reduced by new methods of transportation across Lake Baikal.

- Precautionary measures have been reinforced against fires in the basin of Lake Baikal in order to decrease the number of forest fires and the consequential losses.

- Afforestation has been initiated along the southern coast of Lake Baikal and on the slopes of Khamar-Daban in order to prevent erosion and increase the role of forests in pollution control.

The complex management program of the Baikal region includes rational utilization of the natural resources as well as strict control of the industrial, private and touristic enterprises.

Intense research made in the framework of the scientific project "Siberia" serves as a basis of the realization of this programme.

All these achievements are, however, only one step forward to the more comprehensive protection of the lake. Therefore an additional plan of scientific and economic research has been worked out on the rational use of the natural resources of Lake Baikal. During the last ten years research has been carried out in accordance with this plan.

Of the 50 research institutes and colleges participating in the implementation of this project, 18 belong to the Siberian Branch of the Academy of Sciences of the USSR coordinating the work.

Large-scale studies have been devoted to engineering geology, seismology, hydrology, medical geography and climatology. Water quality and biological resources of the lake as well as utilization of the forests and afforestation in the catchment area were studied thoroughly. New recommendations have been worked out and accepted in order to improve the technology in the paper mills at Baikalsk and Selenga. The results of the five-year comprehensive studies have been synthetized and a proposal has been put forward on the conservation and rational utilization of Lake Baikal.

ANALYSIS OF THE MAIN SOURCES OF POLLUTION

1. More than 250,000 m^3 day^{-1} treated sewage discharge enters the lake from the pulp and paper mill at Baikalsk. In addition, about 150,000 m^3 day^{-1} sewage reaching Lake Baikal is untreated. The chloride loading from the sewage of the mill amounts to 18,000 tons $year^{-1}$, which equals the total loading through the 336 tributaries of Lake Baikal.

Laboratory experiments and field observations of the Limnological Institute, the Biogeographical Institute of the Irkutsk University and the Petrozavodsk University revealed that well treated sewage of the mill at Baikalsk is mutagenic even after a 50- to 100-fold dilution. Changes in the behaviour of the aquatic organisms could be observed when the sewage was diluted 10,000 times. If the sewage of the mill at Baikalsk were diluted to a harmless level, the final volume of the diluted sewage would exceed 2.5 billion m^3 $year^{-1}$.

2. Industrial and municipal sewage discharge of the Ulan-Ude region amounts to more than 120,000 m^{-3} day^{-1}.

Even the diluted industrial sewage has an unfavourable effect in the natural spawning areas of the Baikal omul. In the Ulan-Ude region the mortality of the spawn reaches 98.4%. The main pollutants are fats, fibrous matter, oil products and salts of heavy metals. The surviving individuals of the omul (1.6%) are also exposed to dangerous pollution and produce unhealthy spawn.

3. The pulp and paper mill at Selenga is the principal source of pollution of the Selenga River. Sewage production of this mill amounts to 60,000 m^3 day^{-1}, and the system of the treatment is similar to that in the mill at Baikalsk. The discharge enters the most productive, shallow area of the lake, the so-called Selenga area.

At present, the heavy industrial pollution transported by the Selenga River can be detected even 130 km northeast from the mouth of the river in the lake. The pollution reaches the opposite shore of Lake Baikal towards northwest, 20-25 km from the mouth. The loading transported by the Selenga River could be detected in an area of 200 km^2 within the lake before the construction of the mill at Selenga. By this time this area has increased to about 1500 km^2, and the increasing tendency is still obvious.

The mining works at Dzhydinsk also contribute to the heavy pollution of the tributaries of the Selenga River.

4. Erosion caused by the surface runoff and wind, as well as washout of the soil and toxic substances greatly endanger both the ecosystem and the economy, destroying the ecological balance having been formed during thousands of years.

Due to these processes the area of eroded soils has increased, water discharge of the rivers decreased, dozens of large and hundreds of small rivers have disappeared completely.

According to the data of the Ministry of Agriculture of the Buryat Republic, about 150 rivers and creeks have disappeared by 1982.

According to the data of the Ministry of Agriculture, the area of soils subjected to wind-induced erosion reaches 314,000 ha, including 245,000 ha of fields, 1300 ha of hayfields and 67,900 ha of pastures.

An area of 276,100 ha is eroded by the surface runoff including 215,100 ha of fields, 4600 ha of hayfields and 56,400 ha of pastures. The area of the eroded agricultural soils was 23,800 ha in 1971, increasing ten times and reaching 276,100 ha in 1981.

The area of soils potentially exposed to wind-induced erosion makes up 907,200 ha, including 10,300 ha of hayfields, 135,600 ha of pastures and 761,300 ha of fields. The area of soils potentially exposed to erosion caused by wind and surface runoff increased from 195,300 ha in 1971 to 907,200 ha in 1981, i.e. nearly five times.

The only explanation of the increasing erosion is the uncontrolled deforestation and inconsiderate agricultural land utilization in the catchment area of Lake Baikal. More forests, primarily pine forests, were cut down than planned almost everywhere. In some regions this excess was 5-6 times the planned and under normal conditions would have lasted a whole century. Clear-cutting of the forests resulted in a disruption of the hydrological balance of Lake Baikal and its watershed area.

Due to this disruption as well as to the drier weather during the past ten years, the water level of the lake remained close to the minimum mark. This, in turn, resulted in a decreased outflow through the Angara River, and the capacity of the hydroelectric plant has been reduced to 8-9 x 10^6 kWh $year^{-1}$.

5. Rafting, a means of wood transportation on the lake, is a significant source of pollution. Over 2 x 10^6 m^3 wood is rafted annually on the lake. More than 70,000 tons of organic carbon load originate annually from the washout of bark and wood lost during rafting, increasing the oxygen demand by a

factor of 2.5. This obviously results in a deterioration of the water quality.

6. Atmospheric pollution is particularly harmful for the vegetation in close proximity of the emittors. According to recent research, atmospheric pollution may account for over 32% of the total loading.

Only the mill at Baikalsk emits more than 35,000 tons per year of dangerous substances, including 19 tons day^{-1} of sprays and up to 35 tons day^{-1} of sulphur-containing substances.

Dust and gases emitted by the mill stretch over 160 km towards northeast, reaching the coast of Lake Baikal and 40-50 km westwards threatening the towns of Slyudyanka and Kultuk. The emitted pollution is lifted up to 1500-1800 m spreading over the slopes of Khamar-Daban, reaching the timberline and threatening an aera of 2000 km^2. An area of 160 km^2 of dried forests surrounds the mill at Selenga, and trees with dried shoots can be observed in an area of 600 km^2. This is nearly the same area where the concentration of atmospheric pollutants emitted from different enterprises of Selenga, Guzinoozersk and other towns is the highest.

The lead-zinc foundry at Kholodnensk planned to be built in the region of the Baikal-Amur Railway would heavily pollute the atmosphere in the northern part of the Baikal watershed, as well as in the neighbouring areas.

7. Destruction of the vegetation and the soil along the Baikal-Amur Railway including the protected area of the Baikal watershed resulted in an increased suspended solids load to the lake. In areas where the turfy cover was damaged, the annual loss of soils amounted to 2230 g m^{-2}, while in the areas where the natural cover remained intact, no losses could be observed even during heavy rains.

8. In order to provide the Baikal-Amur Railway construction with different materials, instruments and fuel, the fleet as the main means of transport was remarkably developed on Lake Baikal. The capacity of the fleet increased from 40,000 tons to 1.5×10^6 tons during the period 1975-1981. Water pollution caused by oil products can be traced everywhere, particularly in the harbours and filling stations. Their concentration may

exceed the MPC by a factor of 20-40. When the Baikal-Amur Railway has been completed, this source of pollution was reduced, too.

9. The increasing tourism, especially private boats, may also deteriorate the water quality. Similar to the fleet, private motorboats may discharge oil products both in Lake Baikal and in its watershed area.

Undisciplined groups of tourists are leaving piles of bottles and tins along the coasts. About 80% of the trees have been hewed in some areas. Young trees have been cut down to pitch tents. Several forest fires spring up every year along the routes of such groups.

As a consequence of the destructive pollution in the last decade a decrease could be observed in the growth rate of fishes concerning both their length and weight (Table 1), as well as in their fattiness and fecundity. Maturation of water animals (fish and seal) slowed down. The population growth of the omul has almost doubled during the years when its commercial fishery was banned (1969-1975), with the exception of the so-called Selenga race. The biomass of omul, however, remained unchanged even during this period. The average annual catch of omul amounted to 4960 tons between 1950 and 1962, decreasing to 1320 tons in the periods 1963-1969 and 1975-1982.

During the last 15 years the total annual sewage discharge of the mills at Baikalsk and Selenga and the Ulan-Ude region was about 90×10^6 m^3. The mill at Baikalsk alone emitted about 1.5 billion m^3 industrial sewage during the last 15 years, containing over 800,000 tons of mineral salts and many dangerous substances. Hence, there was an area of 100 km^2 in Lake Baikal where the concentration of pollutants reached the level which may induce mutagenic changes in the water organisms. In addition, about 15,000 km^3 water or half of the total lake volume was contaminated by harmful substances in a concentration which may cause changes in the behavioural responses of the organisms. Since sewage loading enters the littoral zone where the aquatic life is most abundant, the potential damages are realized at their maximum.

Table 1. Weight growth (g) of different age classes of omul populations in Lake Baikal (after V.V. Smirnov 1982)

Year	North-Baikalsk race			Selenga race			Posol race		
	4+	6+	8+	4+	6+	8+	4+	6+	8+
1949	287	357	523	-	397	463	-	517	583
1969	144	318	487	145	261	388	130	262	394
1972	114	323	291	88	143	272	70	124	222
1974	92	184	272	83	161	225	62	95	187
1976	99	174	261	91	153	239	61	122	211
1978	111	193	242	98	169	233	57	127	191
1980	100	192	278	99	167	228	53	117	194
1981	103	189	265	94	158	246	60	107	231

The water balance of Lake Baikal has been seriously disturbed. Deterioration of the ecosystem is a consequence of increased human activity in the watershed of the lake, increased industrial sewage loading, increased use of pesticides and fertilizers both in agriculture and forestry, all of which induce changes in the chemical composition of the surface run-off. The aquatic organisms, having acclimatized to the special environment of Lake Baikal in the long process of evolution, proved to be incapable of adjusting to the recent polluting chemicals. Industrial discharges caused a decrease in the metabolic activities of the aquatic organisms. This is reflected, e.g., in the slower growth rates, decreased fecundity and higher mortality. In 1987, for example, many Baikal seals perished. According to incomplete data, more than 10% of seals perished following the winter kill of sculpins. Based on the number of dead fishes cast ashore by waves and observed along a 20 km coastal section in the southern part of Lake Baikal, their total could be estimated at 20 ind m^{-2}. In the same area mass kill of the invertebrate Macrohectopus branicki was also noticed.

Industrial sewage and dust-and-gas emission although being treated are still harmful for aquatic organisms, and this is the basic problem of the conservation of Lake Baikal. The cardinal solution is to completely eliminate sewage and atmospheric discharges. This is the only possibility not only from the point of view of water management, but also economically, if one takes into account that dilution of the treated but still harmful wastes is much more expensive than the operation costs of the pulp and paper mills.

The main directions of economic and social development of the USSR were outlined at the XXVIIth Congress of the Communist Party of the USSR for the periods 1981-1985 and 1986-1990. The decision stressed the necessary improvement of water management, continuation of lake conservation and the rational utilization of the unique natural resources, first of all that of Lake Baikal.

MEASURES ON THE CONSERVATION OF LAKE BAIKAL

In order to provide normal functioning of the Lake Baikal ecosystem, the following measures are necessary:

1. Complete prevention of any sewage discharge into the lake or its watershed, even if the sewage is treated. The first measure should be rational reorganization of the pulp and paper mill at Baikalsk, e.g. into a wood processing plant without sewage emission. This may largely contribute to the conservation of the unique ecosystem of Lake Baikal. According to the new government decree, the mill at Baikalsk should be reconstructed by 1993.

The purpose of the reconstruction of the mill at Baikalsk is to create a closed water-rotation system and later on to develop a non-waste producing industry. This will promote a marked reduction of pollution reaching Lake Baikal and its watershed.

2. At present, the sewage of the mill at Selenga is discharged into deep horizons of the soil or into the Klyukvenaya Creek. In order to prevent pollution of the groundwater and the soil surface, the mill should be provided with a closed water-rotation system.

3. Discharge of the treated sewage into the Selenga River and that of the most harmful substances into its drainage basin in the region of Ulan-Ude should be stopped.

4. Rafting over the lake should be banned and replaced by shipping.

5. Dust-and-gas wastes should completely be treated in all enterprises in the Baikalsk Basin.

6. More national parks should be established in order to provide suitable areas for growth and breeding of wild plants and animals.

7. Rational management of the lake and its surroundings could be promoted by the establishment of either the "Complex of Baikal" under the authority of the Council of Ministers or the independent, authorized enterprise "Baikal". The legal status of either the "Complex of Baikal" or the "Baikal" should be regulated.

8. Plantation and cutting down of forests should be coordinated and controlled. A more rational forest management should be introduced. Cutting down should be banned in the water-protection zone of Lake Baikal and in its proximity.

9. Agricultural technologies should gradually be replaced by technologies preventing erosion and cattle breeding should preferentially be developed in the Zabaikalsk region of the catchment area of Lake Baikal. The area of low productivity fields should be decreased in arid, waterless regions, since the agricultural value of these sandy and supersandy soils is minimal.

In accordance with the government decrees fish breeding plants should work out new methods and tools for fishery, improve biotechnology of artificial fish breeding and fishery management. Special attention should be paid to the reproduction of omul.

10. Technology of ore processing in the lead-zinc works at Kholodninski should be improved and the ore processing moved outside the catchment area of Lake Baikal in order to prevent sewage discharge into the lake and its tributaries. Sewage production could be decreased if the Dzhidinsk-type polymetallic ores, the Muchor-Talinsk perlites and the Cheremzhansk quartz-

ose sandstone were processed. A series of measures should be taken in order to decrease losses of valuable minerals in the course of mining, processing and transportation.

11. Principles for the establishment of national parks, protected areas, etc. should thoroughly take into account maintenance of the natural equilibrium, conservation of the ecosystem of Lake Baikal as well as the reasonable touristic exploitation of the resources. Uncontrolled tourism should be stopped in the region of Lake Baikal.

A long-term project for the rational exploitation of natural resources of Lake Baikal and its catchment area has been put forward.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

LONG-TERM NATURAL CHANGES IN PLANKTON OF THE SOUTHERN BAIKAL

I.L. RYCHKOV, E.N. KUZEVANOVA, G.I. POMAZKOVA

Research Institute of Biology, Irkutsk State University
Irkutsk, USSR

INTRODUCTION

Due to great disturbances in the ecology of the Baikal region, the estimation and prediction of conditions of the Lake Baikal ecosystem are the most important problems in hydrobiological studies. Such problems may be solved by long-term observations of the lake water communities.

The aim of the present paper is to study the peculiarities of long-term structural changes in the plankton community of the Southern Baikal and to attempt to predict the state of the community considered.

MATERIALS AND METHODS

The collections of the plankton number made at a natural pelagic station No. 1 in Bolshie Koty Bay over the period 1946 to 1986 were included in the study. The station is situated 2 km from the shore and is 800 m deep.

In phytoplankton, the number and biomass of abundant species Melosira baicalensis (=Aulacosira), M. islandica, Stephanodiscus binderanus, Cyclotella minuta, C. baicalensis and the total species number in Synedra, Gymnodinium, Dinobryon, Ankistrodesmus, Anabaena genera were studied. In zooplankton, the number and biomass of the dominating endemic Epischura baicalensis and Cyclops kolensis were examined as well as the total number and biomass of Cladocera and Rotatoria.

The following methods were used to solve the problem: principle components analysis (5), estimation of confidence of long-term trends (11), correlation analysis and linear smoothing (8), analysis of phase diagrams of relative increase rates in annual mean biomass and number (6) and analysis of long-term variations in the percentage of species number and systematic groups in the plankton.

RESULTS AND DISCUSSION

The dynamics of the Southern Baikal plankton is changing considerably from year to year. It is most clearly seen in the cell number of the dominating alga M. baicalensis. Outbreaks in the number of Melosira were observed in 1946, 1950, 1953, 1957, 1960, 1961, 1964 and 1968 which were considered high-yielding years as to Melosira biomass or "melosira" years (2, 3, 4, 9). In the melosira years the positive increase of the annual mean number was always observed not only in M. baicalensis but also in such species as M. islandica, St. binderanus and Synedra species. The abundance of M. baicalensis was not reported in the subsequent years up to 1982, however, the positive increase in their number and accompanying species continued in 1971, 1974, 1976, 1978 and 1979 (4). The authors revealed a long-term decrease in the number of M. baicalensis and an increase in the proportion of more small-cell algae.

The long-term cycles in the dynamics of the total number and biomass were shown to be 16-17 years for phytoplankton and 14 years for zooplankton. The phases of phyto- and zooplankton cycles coincided. Long-term changes of phytoplankton biomass were of the same amplitude in the first and second cycles (Fig. 1a). Yet the general trend of increase, particularly pronounced in 1970-1980, was clearly exhibited along with long-term changes in the phytoplankton number (Fig. 1b). These plots illustrate the process of relative reduction in phytoplankton cell size recorded during the whole period of the study. Following the long-term tendency of decrease the average volume of one cell in the group of abundant algae approached 2000 $m\mu^3$ which is close to the Synedra acus cell size (Fig. 1c). In

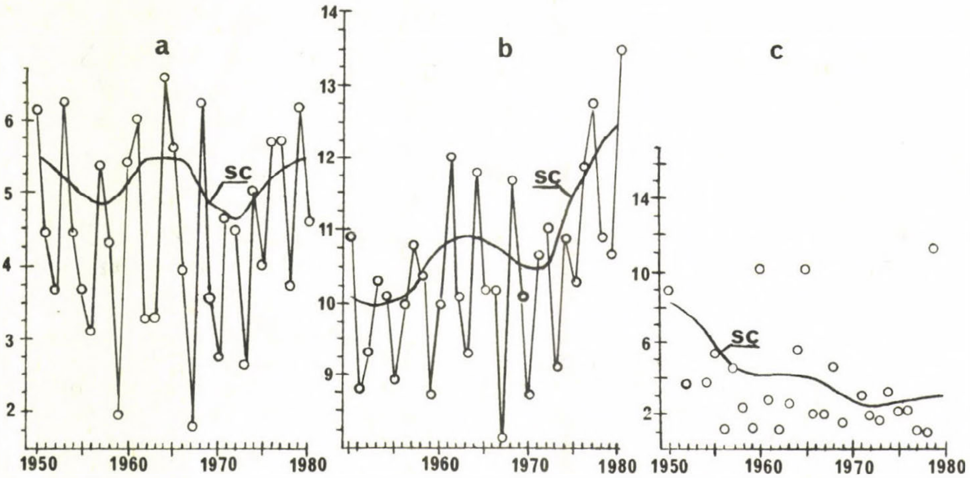


Fig. 1. Long-term dynamics of the average annual biomass (a) and the number of phytoplankton (b). (c) Ratio of biomass to number (average biomass of one cell). Ordinate, natural logarithms of the biomass and number; abscissa, years. sc, smoothed curve (double 7-point smoothing).

contrast to phytoplankton, long-term changes in the total number of zooplankton were registered within a stable long-term average value and its total biomass slightly decreased.

We found three types of seasonal succession in 1946-1971 which were repeated in cycles. The first type of succession was typical of the melosira years. It was characterized by an abundance of M. baicalensis and accompanying algae. The population of E. baicalensis was depressed in this period, while C. kolensis was in more favourable conditions as compared to other years. The second type of succession was found in the next postmelosira years when Synedra species and the corresponding algae predominated and the summer generation of E. baicalensis prevailed. The third type of seasonal succession was characteristic of the premelosira years, i.e. the dominance of C. minuta and accompanying algae, and the primary development of winter-spring generation of E. baicalensis. These types of seasonal succession followed each other from 1950 to 1971 and from 1979 to 1982. Moreover, each type of seasonal succession repeated in 11 years (Fig. 2).

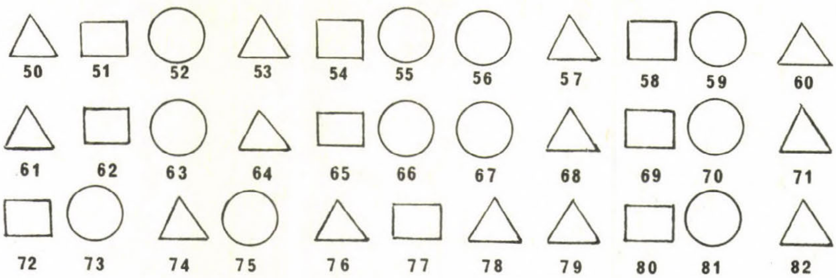


Fig. 2. Scheme of short-term cyclic succession in the plankton of the Southern Baikal. △-melosira years; □-post-melosira years; ○-premelosira years. The numbers below the geometrical symbols designate years (e.g. 66 is 1966)

Also, we have shown different long-term trends in the number of certain planktons to occur along with short-term cyclic successions and long-term changes in the total number and biomass of the phytoplankton. The trends appeared in 1950-1953. The most pronounced long-term changes were demonstrated in the melosira years. The planktons were divided into some groups according to the trend. The most abundant was the plankton group the number of which increased only in the melosira years. The group includes C. minuta, the species of Ankistrodesmus, Anabaena, Dinobryon, Cladocera, E. baicalensis. The number of M. baicalensis and St. binderanus declined in the melosira years. It is interesting that the number of such species as M. islandica (accompanying the development of M. baicalensis) and C. baicalensis (developing together with C. minuta) varies in all three-year types without any pronounced trends within some stable long-term average values. The tendency towards increase in the species number of Synedra genus and N. acicularis is the most dangerous phenomenon as well as the decrease in the number of C. kolensis during the whole period of the study.

Due to different long-term tendencies to the change in plankton number, by 1970-1980 the species structure of the plankton and the quality pattern of its seasonal succession changed and in 1972 the cycle of seasonal succession failed. Thus, in the period considered as melosira according to the scheme, Synedra species predominated over others. In the pre-melosira years the dominance of C. minuta continued, however, the proportion of small-cell species increased significantly. In 1977 and 1980 (postmelosira years), Synedra species were substituted for N. acicularis. The seasonal dynamics in the number of E. baicalensis stabilized in 1970-1980 and repeated from year to year the type of the postmelosira years, i.e. the development of the summer generation prevailed. In 1979-1982 succession was partially restored: in 1979 and 1982 the number of M. baicalensis increased rather markedly and the number of E. baicalensis decreased.

In conclusion, the plankton of the Southern Baikal is characterized by short-term cyclic succession, long-term variations in the number and biomass and long-term trends in the number of certain planktons. Due to such changes in number, by 1970-1980 the short-term cyclic succession failed in the plankton, the dominating species were substituted and in the zooplankton seasonal dynamics stabilized in the dominating species.

Are the changes observed characteristic only of the Southern Baikal? Popovskaya (10) also reported a decrease in the number of M. baicalensis and an increase in the number of small-cell algae not only in the southern but also in the middle and northern basins of the lake. The results, similar to our data on zooplankton, were presented by Afanas'eva (1). Thus, it may be inferred that the long-term dynamics in the number and biomass of the Southern Baikal plankton results from the general processes over the whole water area of Lake Baikal.

REFERENCES

1. Afanas'eva, E.L. Baikal Atlas (in press).
2. Antipova, N.L., Kozhov, M.M. (1953): Materials on seasonal and annual fluctuations in the crop of some widespread forms of phytoplankton in Lake Baikal. Proc. Irkutsk Univ., 7, N 1-2, Biol. Ser., 63-68.
3. Antipova, N.L. (1963): On fluctuations in the number of Melosira species in Baikal plankton. Proc. All-Union Hydrobiol. Soc., 13, 235-241.

4. Ashhepkova, L.Ya., Kozhova, O.M. (1985): Prediction of phytoplankton dynamics in Lake Baikal. In: O.M. Kozhova (ed.): Prediction Methods of Ecological Systems. Nauka, Novosibirsk, pp. 29-56.
5. Dubrov, A.M. (1978): Statistic data analysed by the principle components analysis. Statistica, Moscow, 136 pp.
6. Kuzevanova, E.N. (1985): Phyto- and zooplankton dynamics in Lake Baikal in "melosira" years. In: T.F. Kushnarev (ed.): 3rd Conf. Young Researchers. Irkutsk, 22 pp.
7. Kuzevanova, E.N. (1986): The peculiarity of long-term dynamics of phyto- and zooplankton in the Southern Baikal. Thesis Candidate Degree, Irkutsk, 22 pp.
8. Pollard, J. (1982): Reference book on computing statistic methods. E.M. Chetverikov (ed.): Finansy i Statistika, Moscow, 344 pp.
9. Popovskaya, G.I. (1979): Long-term changes of dominant diatomic algae in pelagic zone of the Southern Baikal. In: O.M. Kozhova (ed.): Ecological Problems of Transbaikal (Abstr. Republ. Conf.). Part 1, Productivity of water ecosystems. Irkutsk, pp. 100-101.
10. Popovskaya, G.I. (1986): The present state and prediction of the development of Baikal phytoplankton. In: G.G. Vinberg (ed.): 5th Cong. All-Union Hydrobiol. Soc. (Abstr.), Part 1. Tol'yatti, pp. 207-208.
11. Zaks, L. (1976). Statistic Estimation. Yu. K. Belyaev (ed.). Statistika, Moscow, 598 pp.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

THE PROBLEMS OF LAKE SEVAN AND WAYS OF SOLUTION

R.O. OGANESIAN, A.S. PARPAROV

Sevan Hydrobiological Station
Kirov 186, Sevan, Armenian SSR, 378610, USSR

Exploitation of water resources has always been a serious problem in the arid Armenia. Lake Sevan is the only big resource of freshwater in Armenia. The outflow was increased artificially by lowering the water level. This, however, caused considerable disturbances in the ecosystem of the lake and finally led to its eutrophication.

Eutrophication of lakes can be described by the following chain of events: increased nutrient input $\rightarrow$ sedimentation $\rightarrow$ recycling of nutrients from the sediments $\rightarrow$ output (Serruya 1974, Martynova 1984).

In most cases eutrophication of lakes is a consequence of a change in the first link of this chain, i.e. the excess nutrient input from the watershed. In this respect Lake Sevan is an exception,since its eutrophication can be related to the disturbed interaction between sedimentation and nutrient recycling from the sediments.

Lake Sevan is the largest water body of the Caucasus, and one of the largest alpine lakes on the Earth. The lake is situated in the northeastern part of Armenia, about 1900 m above sea level. The lake consists of two basins with different morphometry, the deeper Minor- and the shallower Major-Sevan (Fig. 1). Almost all of the tributaries enter Major-Sevan, whereas the outflow, the Rasdan River is from Minor-Sevan. The outflow is regulated so as to meet the demand of the irrigation from May until September. The outflow was artificially increased in 1939 that led to the lowering of the water level from 1916.2 a.s.l. to 1897.7 a.s.l. Average depth and volume of the lake decreased by more than 40% (Table 1). Three stages could be distinguished in the process of lowering the water level (Fig. 1): relatively low rates (0.2 m y^{-1}) were characteristic in the period 1939-1948, high rates (0.9 m y^{-1}) were observed during 1949-1962, and finally the stabilization of the water level occurred after 1964.

The lake was oligotrophic before the reduction of the water level (Table 1). At that time

- the Secchi transparency was high (about 14 m)
- the hypolimnetic oxygen concentration exceeded 4 mg l^{-1} even at the end of the summer stratification
- both algal and macrophyte production was about 50 g C $m^{-2}y^{-1}$, and diatoms dominated the phytoplankton biomass
- bacterial activity was low (Gambaryan 1968).

An essential hydrochemical peculiarity of Lake Sevan was the unusually low ratio of the inorganic forms of nitrogen and phosphorus. The ratio was much less than 1:1. On the other hand, the concentration of orthophosphate-P (0.3 mg l^{-1}) was exceptionally high for an oligotrophic lake.

Eutrophication of Lake Sevan started in the period 1974-1978, when the rate of water level reduction was the highest. The values of the above-listed parameters have become characteristic of eutrophic lakes (Oganesian 1978):

- the Secchi transparency decreased to 2.5-3.0 m
- the hypolimnion became anoxic by the end of summer stratification, concentrations of CH_4 and H_2S may reach 2 mg l^{-1}
- algal production increased considerably amounting to 400-600 mg C $m^{-2}y^{-1}$ (Parparov 1985); production of macrophytes decreased 30 times (Gambaryan 1979); blue-green algal blooms were observed
- bacterial activity increased (Tifenbach 1984)
- secondary production including fish production increased.

In contrast to other eutrophic lakes, concentration of orthophosphate decreased significantly during the eutrophication of Lake Sevan and the ratio of inorganic N:P increased to 3-5 (Parparova 1985).

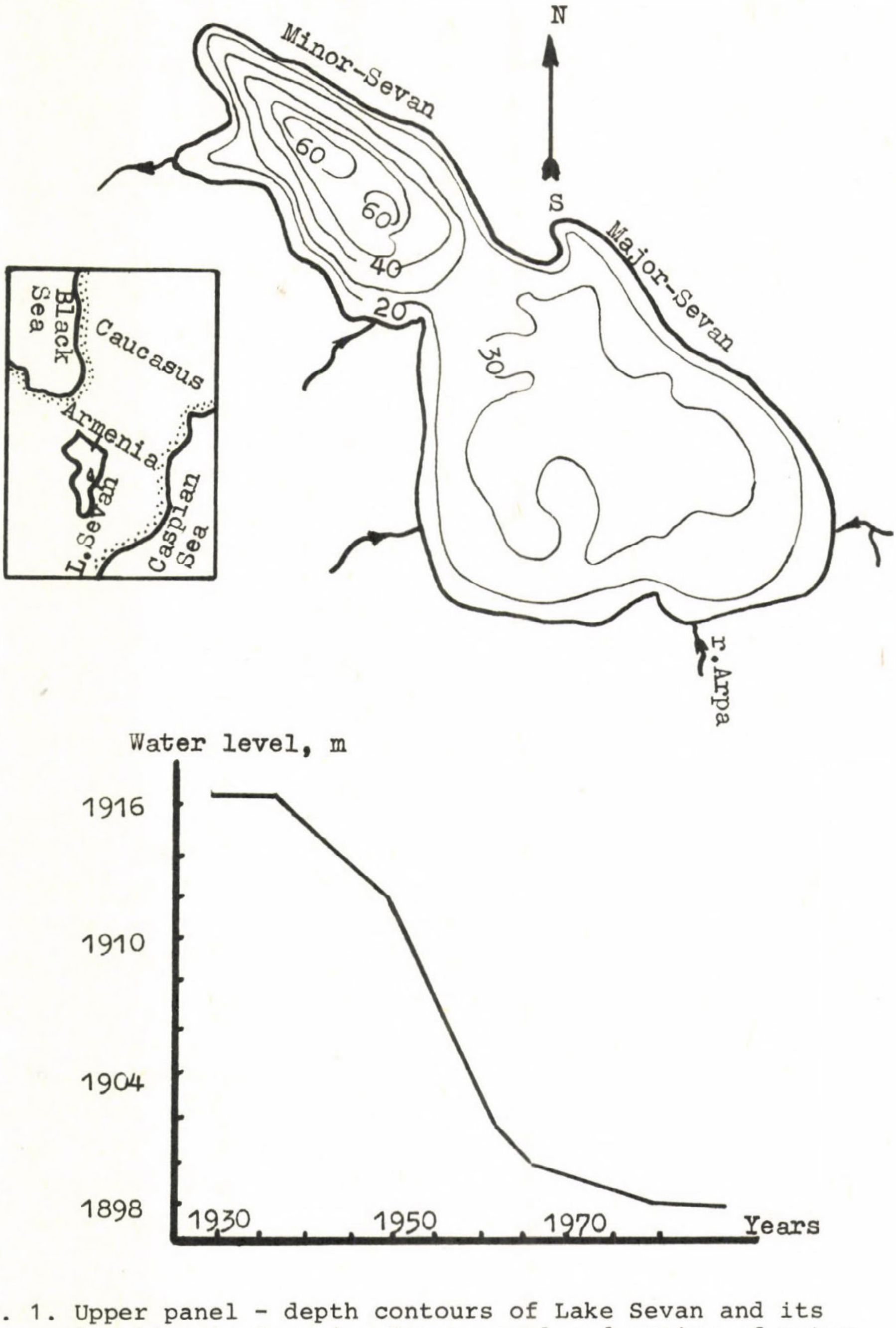


Fig. 1. Upper panel - depth contours of Lake Sevan and its location in Armenia. Lower panel - dynamics of water level lowering

Table 1

Some morphometric and limnological characteristics of Lake Sevan (1 - before the reduction of the water level; 2 - at the present time)

Parameters	Minor-Sevan		Major-Sevan	
	1	2	1	2
Altitude (m)	1916.2	1897.7	1916.2	1897.7
Surface area (km^2)^x	384	328	1032	916
Volume (km^3)^x	19.5	12.8	39.0	29.0
Maximum depth (m)^x	98.8	80.3	58.8	40.3
Average depth (m)^x	50.9	37.8	39.5	22.8
Hypolimnetic O_2 conc. (mg l^{-1})	4.0	0.0	4.0	0.0
pH^{xx}	9.0	8.7	9.0	8.7
Inorganic phosphorus (mg l^{-1})^{xx}	0.32	0.04	0.32	0.04
Inorganic nitrogen (mg l^{-1})^{xx}	0.00	0.10	0.00	0.10
Secchi depth (m)	14.0	3.0	14.0	3.0
Primary prod. (g C m^{-2} y^{-1})	50^{xxx}	250	50^{xxx}	250

^{x} - Gyozalyan (1984); ^{xx} - Parparova (1985); ^{xxx} - estimated by Parparov (1987a)

In 1980-1988 water quality of Lake Sevan improved. Primary production dropped to 250 g C $m^{-2}y^{-1}$, and the blue-green algae were outnumbered by green ones. Production of consumers (including fishes) also diminished. However, at the same time, bacterial production increased in comparison to 1977-1979 (Tifenbach and Parparov 1986).

The elucidation of the mechanism of eutrophication of Lake Sevan is important from the point of view of water management and also because of the unique features of the lake. The limnological research of Lake Sevan revealed that the internal processes played a particularly significant role in its eutrophication.

1. The concentration of phosphorus has decreased for several years. This suggests that the output of this nutrient exceeded its input. A detailed phosphorus budget is shown in Table 2. An excess of phosphorus sedimentation could be esti-

Table 2

Phosphorus budget of Lake Sevan (tons y^{-1}) in 1982–1983 (Parparova 1985)

INPUT		OUTPUT	
Tributary inflow	180	Outflow	20
Seepage	10	Removed by fishes	10
Atmospheric	70	Sedimentation	2700
Diffusive flux from the sediments	960		
Total	1220		2730

mated which was not compensated by increased phosphorus input from the watershed. Direct observations also support this conclusion (Parparova 1985).

2. The nitrogen budget of the lake is significantly different (Table 3). As a result of the economic development in the watershed, the external nitrogen load increased more than the phosphorus load. The share of the inflow is relatively high within the total input. Nevertheless, internal components of the nitrogen budget (sedimentation, recycling from the sediments and nitrogen fixation) are also significant. Evidently, nitrogen fixation was particularly high during the periods of intense blue-green algal blooms.

Table 3

Nitrogen budget of Lake Sevan (tons y^{-1}) in 1982–1983 (Babayan 1984, Parparova 1985)

INPUT		OUTPUT	
Tributary inflow	4000	Outflow	440
Seepage	200	Seepage	60
Atmospheric	1400	Sedimentation	7000
N_2-fixation	4000	Denitrification	130
Diffusive flux from the sediments	9700		
Total	19300		7630

3. Development of anoxic conditions in the hypolimnion is one of the worst consequences of the eutrophication. There is a positive correlation between the rate of hypolimnetic oxygen depletion and primary production in Lake Sevan (Parparov 1987a). According to experimental data, less than 50% of the areal oxygen deficit could be explained by the oxidation of the sedimenting organic matter and the oxygen consumption of the bottom sediments determined by the method of Misandrontsev (1983) (Parparov 1987b).

Consequently, the increase in the rate of oxygen uptake by sediments during the eutrophication of Lake Sevan was not caused only by the increased sedimentation of the organic matter from the euphotic zone followed by its aerobic mineralization in the bottom sediments. One can assume that the increased recycling of the poorly mineralized organic matter having been accumulated in the sediments due to enhanced resuspension and diffusion contributes significantly to the development of anoxic conditions in the hypolimnion.

4. The reduced gases (methane and hydrogen sulphide) released from the sediments may be used for chemosynthesis. The rate of chemosynthetic organic matter production is close to that of the algal production in Lake Sevan. This was estimated on the basis of the absolute values of the dark CO_2 assimilation and the high ratio (up to 40) of dark assimilation to decomposition of the organic matter (Tifenbach and Parparov 1986).

5. Data on the sedimentation show that resuspension of the suspended solids, particulated carbon and particulated phosphorus is an important process in Lake Sevan (Glooshchenko 1988, Parparova 1985).

The rate of resuspension increased continuously with the lowering of the water level. This has also been confirmed indirectly by calculating the Fee index, S_{be}/V_e, (Fee 1979), which is equal to the change in the ratio of the unit bottom area in contact with the epilimnion (S_{be}, m^2) to the unit volume of the epilimnion (V_e, m^3) caused by one meter reduction of the water level (Fig. 2, Oganesian 1987).

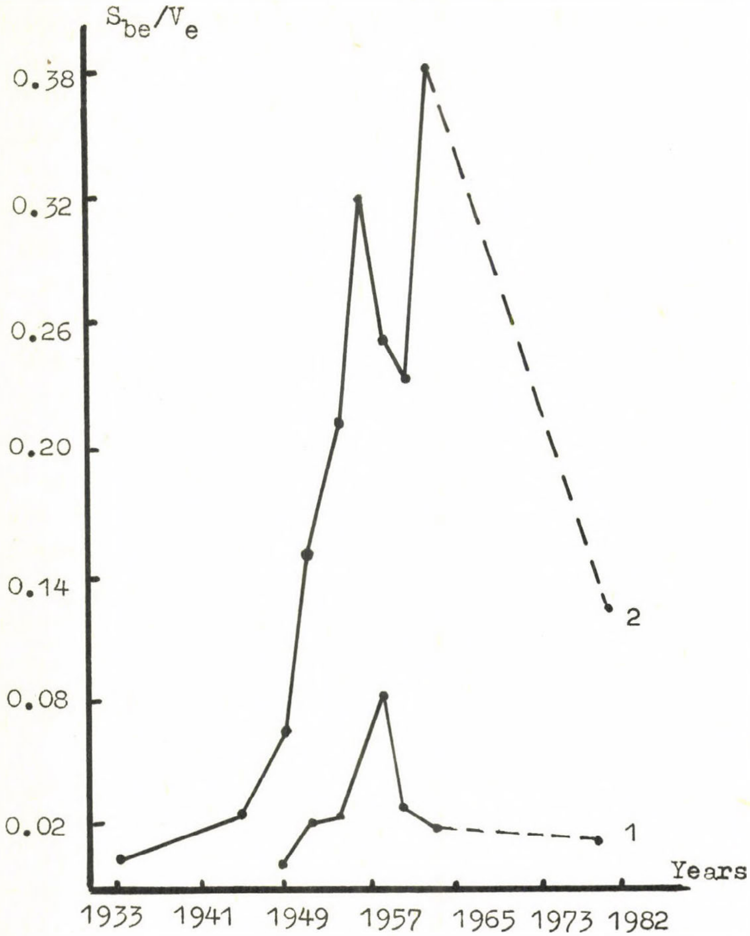


Fig. 2. Dynamics of Fee index in Lake Sevan (see text).
1 - Minor-Sevan; 2 - Major-Sevan

6. Due to the diminishing volume of the hypolimnion, the temperature of the bottom layers increased. This may lead to an increased diffusive release of the nutrients from the bottom sediments. This process is particularly significant in the shallow Major-Sevan. Stratification became unstable in this part of the lake resulting in an alteration of oxic and anoxic periods. Nutrient release was also enhanced.

7. As a consequence of the eutrophication, the species composition of the macrozoobenthos community changed remarkably. The total biomass of the macrozoobenthos increased by a factor of six, including a 30-fold increase of the biomass of chironomids in the most productive years (Ostrofsky 1984). Taking into account the role of chironomids in nutrient release from the bottom sediments (Martynova 1984), it is evident that their increased biomass contributes significantly to the enhanced nutrient recycling.

It is clear that many aspects of the eutrophication of Lake Sevan are related to one of the key problems of current limnology, i.e. the interaction between water and sediments.

The influence of the watershed cannot be ignored. Stability of the lake with respect to the effects from the watershed decreased considerably, as it can be seen from the diminishing values of the index of Kerekes (1974) (Fig. 3). The development of economy in the watershed resulted in an elevated external load of phosphorus and particularly nitrogen.

In Lake Sevan the increased external and internal nutrient loadings contributed to the eutrophication in the same degree (Oganesian and Parparov 1983).

This way an effective eutrophication management in Lake Sevan should include measures which aim at reducing the availability of nutrients released from the sediments in addition to measures which aim at increasing the stability of the lake with respect to the effects of the watershed.

One of the most crucial measures is to increase artificially the mean depth of the lake by introducing nutrient-poor water into the lake. This way the external loading can also be decreased.

In ecological terms raising of the water level has the same consequences as removal of the bottom sediments. The latter is one of the most radical ways of recovering the lakes (Janus and Vollenweider 1985), but cannot be used in large lakes such as Lake Sevan.

The Armenian government adopted a resolution on introducing the Arpa River into Lake Sevan. In connection with this

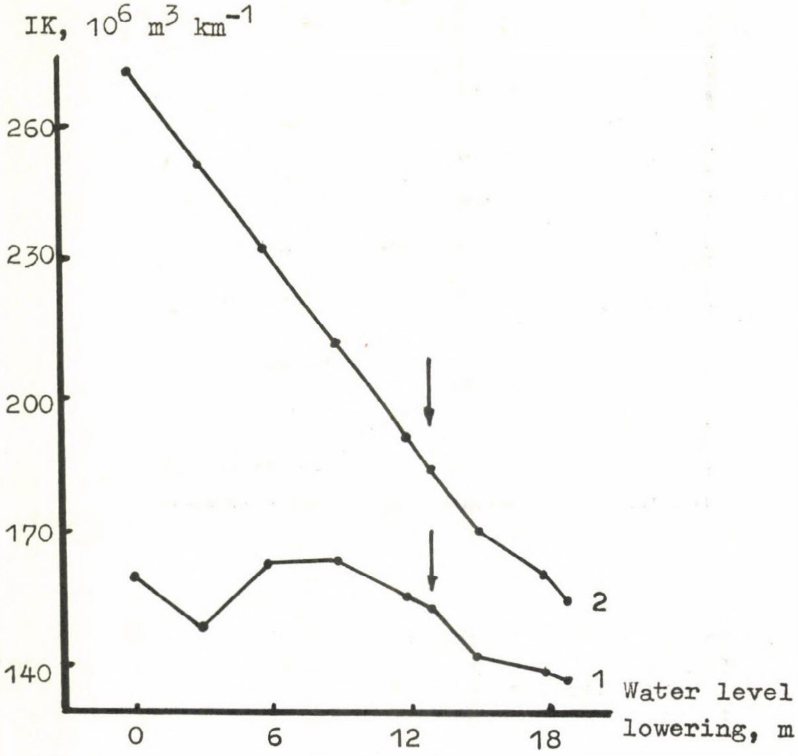


Fig. 3. Dynamics of the Index of Kerekes (IK) during the water level lowering of Lake Sevan

and additional management measures, research should be carried out in order to determine the optimal water level of the lake, as well as to predict its ecological state.

The optimal water level of the lake is not only a limnological, but also a socio-economic question. From the point of view of hydroecology, a minimal rise in the water level would be enough to achieve a significant water quality improvement. Due to the unique features of the lake, the causes of its eutrophication and the special management measures (artificial changing of its morphometry), traditional methods of hydrobiological prognostication cannot be applied, including the loading models of Vollenweider or Dillon and Rigler. A possible

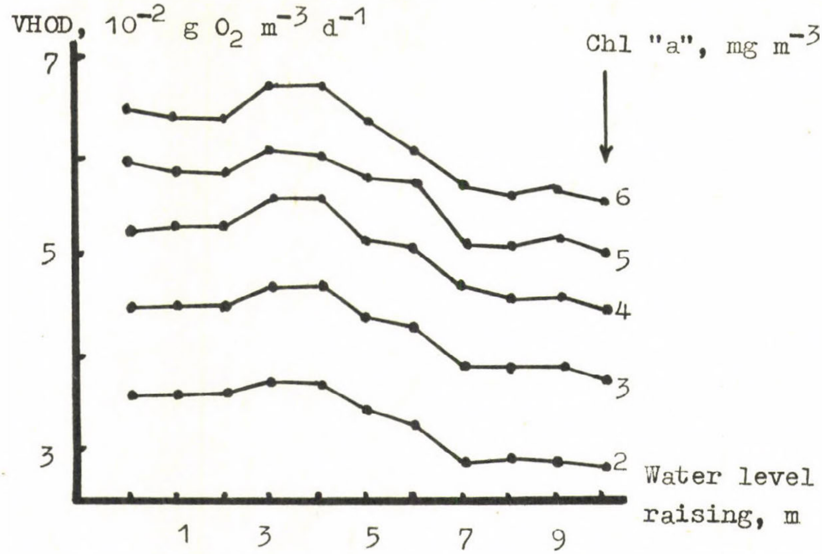


Fig. 4. Chlorophyll "a" concentration (marked by arrow) as a function of volumetric hypolimnetic oxygen deficit (VHOD) calculated according to Charlton (1980) at different values of water level raising

approach is to find the extremes of the relations between morphometric and productivity indexes of the lake. Many of these functions are monotonous.

There is a complicated relationship between the volumetric oxygen deficit calculated according to Charlton (1980) and the changes in the chlorophyll-a concentration during raising the water level of the lake (Fig. 4). Analysis of this and other relationships shows that a significant water quality improvement could be achieved in Lake Sevan raising its water level by minimum 6 meters. Subsequently, the stability of the lake would increase considerably (see index of Kerekes, Fig.3, marked by arrows), together with a decrease in the oxygen deficit (Fig. 4).

Summarising our data obtained in a large alpine lake, Lake Sevan, which has undergone a eutrophication as a result of the artificial decrease of its water level, we come to the following conclusions:

- water level lowering has proved to be an unusual mechanism of eutrophication changing the cycles of nutrients within the lake in an analogous way to the well-known examples of eutrophication in other lakes where the reason is the increased external nutrient loading
- changing the morphometry (mean depth) of a lake may be as efficient eutrophication management measure as changing the external loading.

References

Babayan, J.K. 1984. Microbiological processes in the nitrogen cycle of Lake Sevan (in Russian). Autoref.diss., Moscow, 25 p.

Charlton, M.N. 1980. Hypolimnion oxygen consumption in lakes: discussion of productivity and morphometry effects. Can.J.Fish.Aquat.Sci. 37: 1531-1539.

Fee, E.J. 1979. A predictive relation between lake morphometry and primary productivity and its use in interpreting whole lake experiments. Limnol.Oceanogr. 24: 401-416.

Gambaryan, M.Ye. 1968. Microbiological studies in Lake Sevan (in Russian). Yerevan, 166 p.

Gambaryan, P.P. 1979. Distribution of macrophytes in Lake Sevan. In: Ecology of hydrobionts in Lake Sevan (in Russian). Yerevan, pp. 123-129.

Glooshchenko, L.O. 1988. Structure and role of seston in the ecosystem of Lake Sevan (in Russian). Autoref.diss. Minsk, 24 p.

Janus, L.L. and R.A. Vollenweider. 1985. Phosphorus residence time in relation to trophic conditions in lakes. Verh. Int.Ver.Limnol. 22: 179-184.

Kerekes, J. 1974. The index of lake basin permanence. Int. Rev.Ges.Hydrobiol. 62: 291-293.

Martynova, M.V. 1984. Nitrogen and phosphorus in the bottom sediments of lakes and reservoirs (in Russian). Nauka, Moscow, 142 p.

Misandrontsev, I.B. 1983. Bottom sediments. In: Compartments of the ecosystem of Lake Baikal (in Russian). Novosibirsk, pp. 46-99.

Oganesian, R.O. 1978. Disturbances in the water-sediment interaction as one of the causes of eutrophication in Lake Sevan. In: Water-sediment interactions (in Russian). Yerevan, pp. 43-47.

Oganesian, R.O. and A.S.Parparov. 1983. Ecological aspects of the problems in Lake Sevan. In: Productivity of the ecosystem of Lake Sevan (in Russian). Yerevan, pp. 5-13.

Ostrofsky, I.S. 1984. Productivity of the dominant species of zoobenthos and their role in the ecosystem of Lake Sevan (in Russian). Autoref.diss. Leningrad, 24 p.

Parparov, A.S. 1983. Some aspects of algal production in Lake Sevan. In: Productivity of the ecosystem of Lake Sevan (in Russian). Yerevan, pp. 14-50.

Parparov, A.S. 1987a. Characteristics of phytoplankton production in Lake Sevan and its relation with the hypolimnetic oxygen consumption of the lake. In: Hydrobiological study of productivity in aquatic ecosystems (in Russian). Nauka, Leningrad, pp. 82-90.

Parparov, A.S. 1987b. Estimation of the rate of hypolimnetic oxygen depletion in Lake Sevan. In: Water-sediment interactions (in Russian). Yerevan, pp. 102-106.

Parparova, R.M. 1985. Peculiarities of phosphorus cycle in Lake Sevan with respect to changes in the hydrochemical regime of the lake (in Russian). Autoref.diss., Rostov-on-Don, 24 p.

Serruya, C. 1974. Nitrogen and phosphorus balances and load biomass relationship in Lake Kinneret (Israel). Verh.Int. Ver.Limnol. 19: 1357-1369.

Tifenbach, O.J. 1984. Bacterial flora and bacterial production under present hydrobiological regime of Lake Sevan (in Russian). Autoref.diss., Minsk, 24 p.

Tifenbach, O.J. and A.S.Parparov. 1986. Interrelation between bacterial production and production-decomposition processes in Lake Sevan (in Russian). Proc.5th Congress of Hydrobiol.Soc., Kuibyshev, 1: 216-217.

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

TWIN LAKES, COLORADO, U.S.A.: ECOLOGICAL STUDIES OF THE EFFECTS OF PUMPED-STORAGE HYDROELECTRIC DEVELOPMENT ON A PAIR OF MONTANE LAKES

J.J. SARTORIS

U.S. Bureau of Reclamation, Denver, Colorado, U.S.A.

Twin Lakes are located on the eastern slope of the Rocky Mountains in western North America, at 39° 05' N. latitude and 106° 20' W. longitude. They lie at an altitude of 2804 m above mean sea level, in the Montane Life Zone of Colorado. Twin Lakes are a pair of connected lakes which were originally formed by the morainal damming of Lake Creek by a Pleistocene glacier (figure 1). This creek, a small tributary of the Arkansas River, was the main source of inflow to the lakes in their natural state.

The Lake Creek watershed is relatively small, with an area of only 238 km^2. While the lakes themselves lie in deposits of glacial till, Lake Creek drains mainly the crystalline rocks and eroded ore bodies of the eastern slope of the Sawatch Range, along the Continental Divide. Consequently, the deeper sediments of both lakes, which are composed of a fine glacial rock flour, have accumulated large quantities of heavy metals, including iron, manganese, copper, zinc, lead, and cadmium.

The climate here is cool and semi-arid (table 1). Twin Lakes are dimictic, with an ice cover usually lasting from early December to early May in an average year. Maximum ice thickness ranges from about 60 cm, in an average year, to a meter or more during severe winters. Summer thermal stratification normally reaches its maximum in August, with surface water temperatures in the range of 15° to 18° C, and bottom temperatures of about 9° C.

Twin Lakes were first mentioned in the scientific literature in 1874 (Hayden 1874), and such notable scientists as David Starr Jordan (1891) and Chancey Juday (1906) did brief surveys of the fish and limnology of the lakes around the end of the last century. At that time, the lakes were two distinct bodies of water connected by Lake Creek. Upper Twin

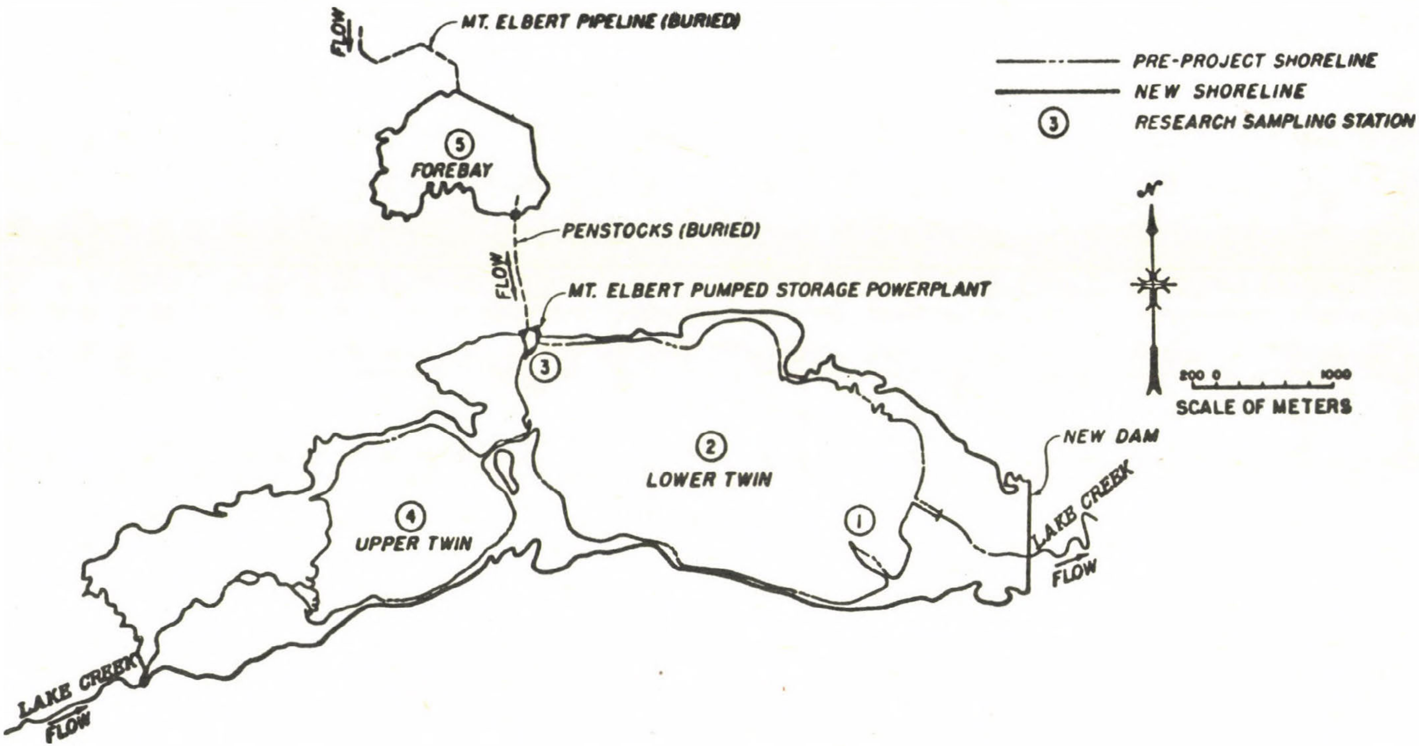


Figure 1 – Twin Lakes, Colorado

Map of the Twin Lakes System

Table 1. - Climatological Data, Leadville, Colorado.

Precipitation:	
Mean annual	469.4 mm
January mean	33.5 mm
July mean	69.3 mm
Mean annual pan evaporation:	1219.2 mm
Temperature:	
Mean annual	2.3 °C
January mean	-7.8 °C
July mean	13.8 °C
Mean frost-free growing season: (June 14 to September 7)	85 days
Solar radiation:	
Mean annual	425 gm-cal/cm^2/day
January mean	240 gm-cal/cm^2/day
July mean	650 gm-cal/cm^2/day

Lakes had a normal surface elevation nearly 2 m higher than the lower lake, and the natural fluctuation of surface elevation in each lake probably did not exceed 0.6 m annually. The normal surface area of the upper lake was about 220 ha, with a corresponding maximum depth of about 25 m. Lower Twin Lakes was larger in surface area, about 610 ha, but more shallow, with a maximum depth of about 23 m. Both lakes were probably oligotrophic, but they harbored diverse zooplankton and fish communities. One species of native trout was described as being endemic to Twin Lakes (Jordan and Evermann 1889).

Permanent settlement began in this area in the late 1870's, and almost immediately the settlers began changing the ecology of Twin Lakes through the introduction of exotic species, and by modifications of the hydraulic regime, which were aimed at converting the lakes into water-supply reservoirs for irrigation and mining. By 1971, when the present studies began, Twin Lakes had settled into a new and different ecological equilibrium.

The lakes were still oligotrophic (mean chlorophyll-a concentration about 3 mg/m^3) and phosphorus-limited. Total dissolved solids averaged about 52 mg/L, and mean hardness, as $CaCO_3$, was approximately 32 mg/L.

The major cation was calcium (10 mg/L), and the major anions were bicarbonate (25 mg/L) and sulfate (14 mg/L). Mean Secchi depth was about 4 m, and pH averaged 7.4.

The fish and zooplankton communities, however, were much less diverse than they had been a century before. Two species of catostomids (suckers) were all that remained of the native fish. The sport fisheries in the lakes consisted of two introduced trout species, the rainbow trout (*Salmo gairdneri*) and the lake trout (*Salvelinus namaycush*). Although the rainbow trout fishery was maintained by annual stocking, the lake trout fishery was self-sustaining, thanks to another species introduction: the opossum shrimp (*Mysis relicta*). The shrimp, on the other hand, had virtually eliminated cladocerans from the zooplankton community, which now consisted largely of two copepod species.

Since the turn of the century, a series of hydraulic engineering works had converted Twin Lakes into a pair of connected reservoirs. First, the natural outlet of the lower lake was dammed, and a deeper, gated outlet was constructed about 1 km to the north. This arrangement allowed a potential vertical fluctuation of 7.8 m in the water surface elevation of the lower lake. Next, the stream connecting the lakes was dredged into a channel that allowed the two bodies of water to fluctuate essentially as one. Finally, a tunnel was constructed under the Continental Divide to divert water from the western slope into Lake Creek. This transmountain diversion increased the total annual discharge of Lake Creek by an average of 42%. It is important to note here that this additional inflow was all added during the irrigation season, which extends from approximately June through September. Thus, the extra flow had the effect of increasing and prolonging the natural runoff peak in Lake Creek.

The net result of these hydraulic changes was increased erosion in the inflow area of the upper lake. What had originally been a marshy meadow was now an eroded floodplain, and the resulting woody debris was deposited in the bottom of Upper Twin Lakes. During severe winters, with prolonged ice and snow cover, this allochthonous organic deposit exerted a biochemical oxygen demand that resulted in extreme hypolimnetic anoxia, with toxic metal releases from the sediments. We documented one such incident of "winter kill" in April 1975 (Sartoris et al. 1977), and it took at least two years for the plankton and benthos of the upper lake to recover to their former levels of diversity and abundance. Primary production in the upper lake was also limited by intense flushing and turbidity during the spring runoff period. In fact, the upper lake functioned as a settling basin and buffer for the lower lake.

The impetus for our studies at Twin Lakes was the Bureau of Reclamation's plan to include the lakes in its Fryingpan-Arkansas Project. This is a very large transmountain diversion project designed to bring water from the Upper Colorado River Basin into the Upper Arkansas River to augment irrigation and municipal water supplies in eastern Colorado. Twin Lakes were included in the project as the site of a pumped-storage powerplant, which was intended to produce peaking power and thus help defray the costs of the entire project. At Twin Lakes, the project features include a 200-MW pumped-storage hydroelectric plant on the northwest shore of the lower lake and a new dam downstream of the old lower lake outlet works (figure 1). The powerplant also includes a constructed forebay, located on the ridge to the north of Lower Twin Lakes. Project water enters this forebay through the Mt. Elbert Conduit from Turquoise Reservoir, which is located 17 km north of Twin Lakes. Turquoise Reservoir is the eastern terminus of the tunnel bringing the diverted water from the western slope of the Continental Divide.

Construction of the Mt. Elbert Pumped-Storage Powerplant began in 1971, and the first 100-MW unit became operational in 1981. The second unit went "on-line" in 1984. Closure of the new Twin Lakes Dam in late 1983 raised the maximum water surface elevation in both lakes by 2.2 m. This had the effect of transforming the two connected impoundments into a single reservoir with two distinct basins. Our studies, and those of our cooperators from the Colorado Division of Wildlife and the Cooperative Fishery and Wildlife Research Unit at Colorado State University, were designed to assess the effects of this pumped-storage development on the ecology of Twin Lakes. The combined studies began in 1971 and ended in 1986.

The effects we observed can be grouped under three general headings, as being related to:

1.) raised lake levels behind the new dam,

2.) increased inflow to the lakes from the Mt. Elbert Conduit, and

3.) pumping and generating operations at the powerplant.

Raised lake levels have changed the morphometry of the lakes, and in particular, the morphometry of the upper lake (table 2). Maximum surface area of Upper Twin Lakes was increased by 45%, while maximum volume

Table 2. - Morphometric Data, Twin Lakes, Colorado.

	Pre-Project Conditions			Present Conditions		
	Upper lake	Lower lake	Both lakes	Upper lake	Lower lake	Both lakes
Maximum water surface elevation (meters)	-	-	2802	-	-	2804
Maximum surface area (hectares)	263	737	1000	381	742	1123
Maximum depth (meters)	28.0	27.1	-	30.2	29.3	-
Maximum volume (cubic meters)	4.11×10^7	11.26×10^7	15.37×10^7	5.53×10^7	11.84×10^7	17.37×10^7
Mean depth (meters)	15.6	15.3	-	14.5	16.0	-
Shoreline length (kilometers)	6.3	10.8	15.9	10.7	13.7	24.4
Shoreline development index	1.09	1.12	-	1.54	1.42	-

increased by 34%, but the mean depth actually **decreased** by 7%. This reflects the fact that most of the lake's expansion was into the shallow western floodplain. In general, the bathymetry of the upper lake basin changed from a roughly circular shape to a more oblong configuration, with the deep end near the connecting channel. This new shape has altered the pattern of summer inflow to the lake. Formerly, the cooler water of Lake Creek entered Upper Twin Lakes as a plunging inflow that cooled and oxygenated the hypolimnion. Now warming in the extensive shallow area causes the creek water to enter the pelagic area of the lake as a near-surface inflow, warming and mixing the epilimnion. The flooded willows and other riparian vegetation seem to be serving as new habitat for fish, zooplankton, and shrimp, and the area may also be acting as a sediment trap and nutrient buffer for the main body of the lake.

In both lakes, the inundation of new terrain resulted in a "trophic upsurge", typical of new reservoirs. Total phosphorus (figure 2) and nitrate (figure 3) concentrations increased significantly in both lakes,

Figure 2 — Total Phosphorus

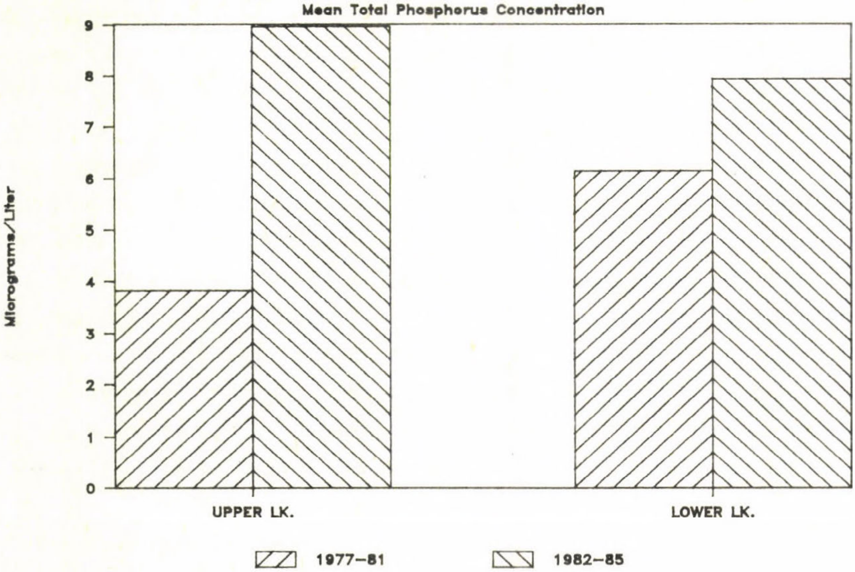
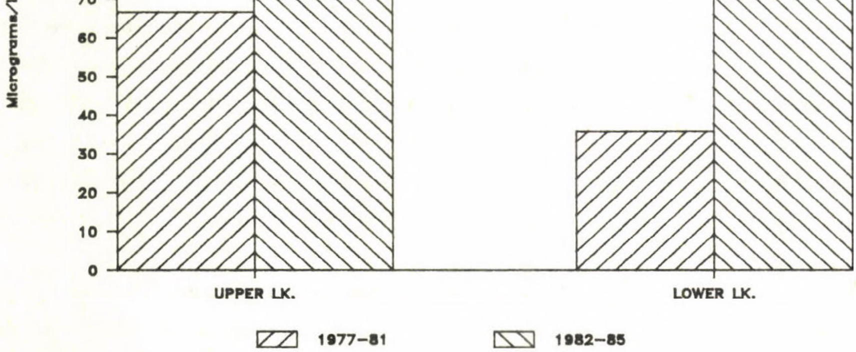


Figure 3 — Nitrate



and chlorophyll-a concentrations increased in the lower lake. No significant increase in primary production was noted in the upper lake, but this could be due to the fact that we failed to extend our sampling efforts into the newly inundated shallow area, which may be the locus of any productivity increase.

The effects of increased inflow caused by the importation of Fryingpan-Arkansas Project water have been almost entirely confined to Lower Twin Lakes (figure 4). At the conclusion of our studies, this added inflow had not affected hydraulic residence time in the lower lake, because it was offset by the increase in lake volume. However, this situation has probably changed with the recent addition of water from a second, municipal water diversion project to the Fryingpan-Arkansas system. An expansion of this municipal project is now proposed, so even greater inflows, and flushing, may be expected for Lower Twin Lakes in the future.

Water imported by the Fryingpan-Arkansas Project is even more soft and dilute than that of Twin Lakes, which has resulted in a significant dilution of the lower lake (figure 5). Mean total dissolved solids concentration of this lake declined to about 40 mg/L, with $CaCO_3$ hardness averaging 27 mg/L. Thus, Lower Twin Lakes is now even less chemically buffered than it was previously, and, therefore, it is more vulnerable to toxicity if heavy metals were to be released from the sediments or introduced with the inflow.

Pumping and generating operations at the Mt. Elbert Powerplant affect both lakes in that circulation of water is increased. During generation, flow direction in the channel connecting the two lakes is often reversed. The main impact, however, falls on the lower lake. Strength and duration of summer thermal stratification in this lake is decreased by powerplant operation (figure 6). Light penetration has also decreased somewhat in both lakes (figure 7), due partly to stirring of fine sediments and to shore erosion caused by fluctuating water levels. Winter ice cover has been weakened around the edges of the lake, again due to water level fluctuation, and the tailrace and connecting channel remain ice-free during powerplant operation.

Of course, entrainment of fish and shrimp during pumping operations is of major concern. Mortality of entrained fish is estimated to average 62%, but the various species differ in their vulnerability to

Figure 4 – Annual Inflow Volume

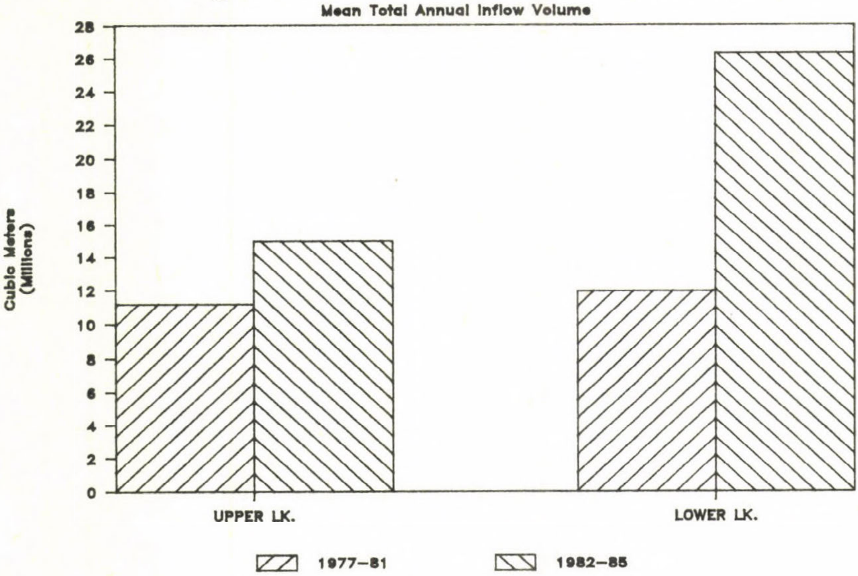


Figure 5 – Conductivity

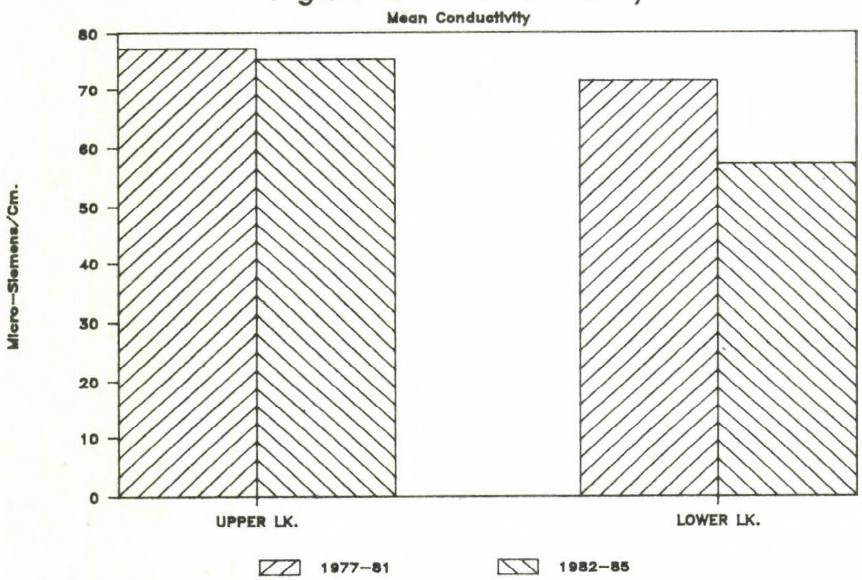


Figure 6 — Thermal Stratification

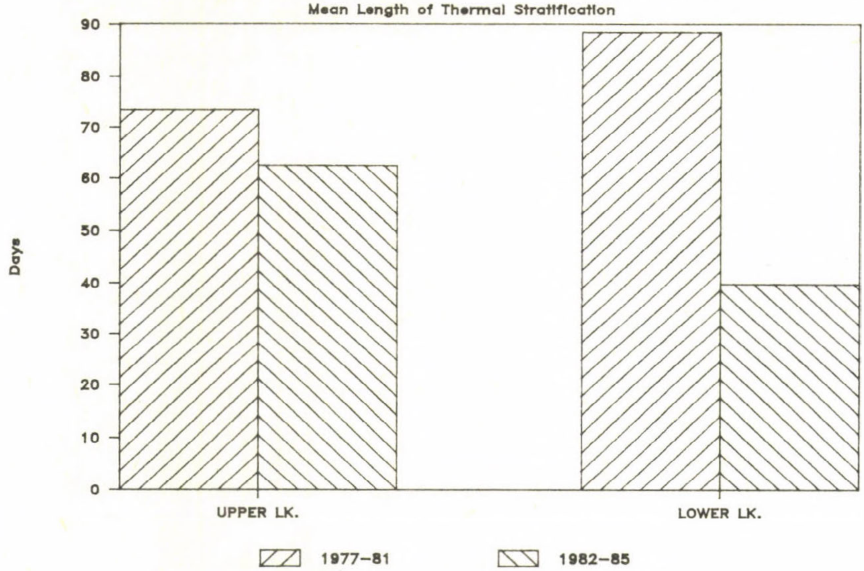
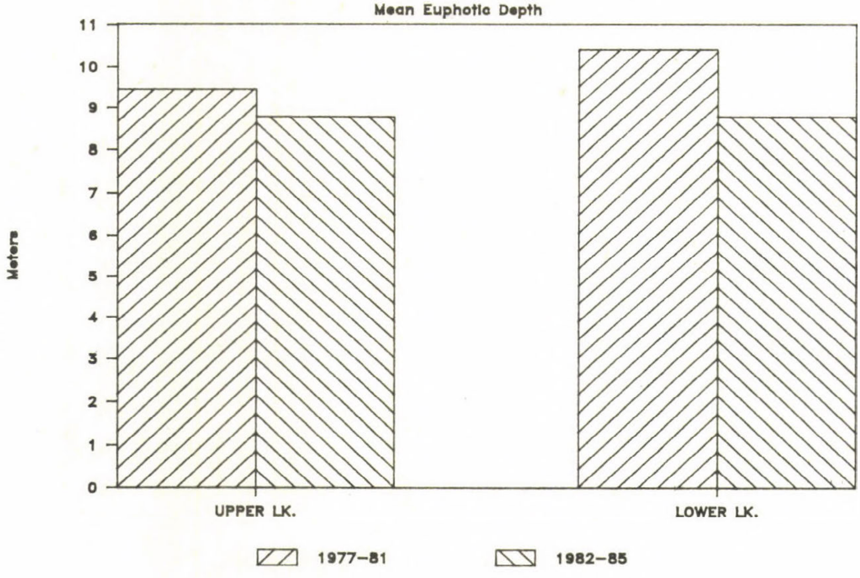


Figure 7 — Euphotic Depth



entrainment. Lake trout and rainbow trout appear to avoid pumping flows, while Kokanee salmon (Oncorhynchus nerka), which were recently introduced, have a high entrainment rate. Mysis shrimp are entrained in significant numbers during nighttime pumping operations, because they migrate to the surface during hours of darkness. Operating criteria for the Mt. Elbert Powerplant call for adjusting pumping schedules to avoid hours of maximum shrimp vulnerability. Mortality of entrained shrimp is less than that of fish, however, and a significant shrimp population has been established in the powerplant forebay. Finally, although zooplankton were not addressed in the powerplant entrainment studies, there is some evidence that lower lake zooplankton populations may be experiencing entrainment impacts similar to those affecting the Mysis population.

I'll conclude this presentation with a brief and speculative look at the possible future equilibrium state of Twin Lakes.

We can expect the "trophic upsurge" in the lakes to level off or decline somewhat in the next few years, as new habitats and niches are filled, and easily leached nutrients are used up. Shoreline erosion was already subsiding in 1985, as new beaches and benches were forming.

Upper Twin Lakes is now the less flushed of the two, and the extensive new shallow inflow area should diminish the impacts of spring runoff a great deal. We also have some evidence that circulation induced by powerplant operation may lessen the degree of winter hypolimnetic anoxia in the upper lake. All of these factors could combine to eventually make Upper Twin Lakes the more productive of the two, and a refugium for fish and shrimp populations.

Lower Twin Lakes, on the other hand, seems destined to become an intensely flushed and manipulated reservoir. Entrainment, flushing, and the effects of powerplant-induced circulation will probably limit its productivity to a lower level than that of the upper lake.

Researchers from the Bureau of Reclamation, the Colorado Cooperative Fishery and Wildlife Research Unit, and the Colorado Division of Wildlife are presently engaged in writing a comprehensive monograph on the Twin Lakes ecological studies. We hope to have this monograph published within the next two years.

REFERENCES

Hayden, F.V. (1874): Annual Report of the United States Geological and Geographical Survey of the Territories for 1873. U.S. Government Printing Office, Washington, D.C., pp.47-54.

Jordan, D.S. (1891): Report of explorations in Colorado and Utah during the summer of 1889. Bulletin of the United States Fish Commission, V.9, pp.1-40.

Jordan, D.S. and Evermann, B.W. (1889): Description of the yellow-finned trout of Twin Lakes, Colorado. Proceedings of the National Museum, V.12, N.780, pp.453-454.

Juday, C. (1906): A study of Twin Lakes, Colorado. Bulletin of the Bureau of Fisheries, V.26, N.616, pp.147-178.

Sartoris, J.J., LaBounty, J.F. and Newkirk, H.D. (1977): Historical, Physical, and Chemical Limnology of Twin Lakes, Colorado. Laboratory Report REC-ERC-77-13, U.S. Bureau of Reclamation, Denver, CO., 86 pp.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest
Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

INFLUENCE OF HURRICANE KATE ON THE WATER QUALITY OF LAGUNA DE LA LECHE, CUBA

R. GÓMEZ CARRO,[1] M.C. PÉREZ EIRÍZ,[2] M.A. PUBILLONES,[3] C. PAZOS[4]

[1]Provincial Commission for Environmental Protection of Ciego de Avila, Cuba; [2]Institute of Ecology and Systematics, Cuban Academy of Sciences; [3]Institute of Hydroeconomy of Cuba; [4]Meteorological Institute, Cuban Academy of Sciences

Abstract

The effects of Hurricane Kate on the water quality of Laguna de la Leche were followed by three samplings. The decrease in the BOD, COD, dissolved oxygen, pH and alkalinity indicated that decomposition of the dragged along organic material increased after the passage of the hurricane.

Enhanced decomposition of the organic matter resulted in an increased regeneration of mineral nutrients, such as ammonia and phosphate. This was followed by an explosive algal development with notable increase in pH and concentration of dissolved oxygen and a decrease in the values of BOD and COD.

Our data indicate a dilution of the water resulting in a decrease of the total number of coliform and fecal organisms.

We can conclude that the passage of Hurricane Kate has improved the water quality of the lagoon. This conclusion is based on the decreased number of coliform organisms as well as on the intense phyto- and zooplankton production observed during March 1986.

Introduction

It is well known that tropical hurricanes are the strongest storms of the world. The wind and precipitation associated with these meteorological phenomena bring about changes in physical and chemical characteristics of the water bodies with direct and indirect effects on the processes of biocoenosis. In addition to this, the hurricanes may cause considerable damages in the tropical regions.

On 19 November 1985 approximately from 1.00 to 1.30 a.m. Hurricane Kate passed over most of the northern coast of the Republic of Cuba, and its center damaged the outskirts of Morón City in the proximity of Laguna de la Leche.

The aim of this study is to analyse the effects of Hurricane Kate on the water quality of Laguna de la Leche, as well as to evaluate the process of its recovery.

Environmental effects of tropical cyclones were studied in the sea (1,2,3) including the impacts of storms on corals, the plankton, fishes and seaweeds (4,5,6,7). The effects of cyclones are also known in freshwater ecosystems (8). The effects of monsoons were studied in Indian dams (9,10), whereas those of the heavy rains in Annie's Lake, Florida (11). However, changes of the water quality caused by a tropical cyclone have not been studied yet in a coastal lagoon.

Materials and methods

Laguna de la Leche situated northward of Morón City is a shallow, brackish, heavily polluted coastal lake in Cuba, connected with the sea through the Chicola Channel (Fig. 1). Its morphometric parameters are given in Table 1. Point sources of pollution are shown in Figure 1.

Samples were taken three times: 13 days before the arrival of the hurricane (6 November 1985), as well as 30 and 125 days after the passage of the cyclone (19 December 1985 and 24 March 1986, respectively). Stations 1,2,4,6, and 9 represent heavily polluted streams. The other stations are located in less polluted areas of the lagoon (Fig. 1).

Physical and chemical analysis was performed according to the standard methods of water and waste-water analysis (12). The number of total and fecal coliforms was determined using the multiple fermentation test tube technique (12). Phyto- and zooplankton samples were analysed according to Javornicky (1958) and Straškraba and Hrbacek (1966), respectively.

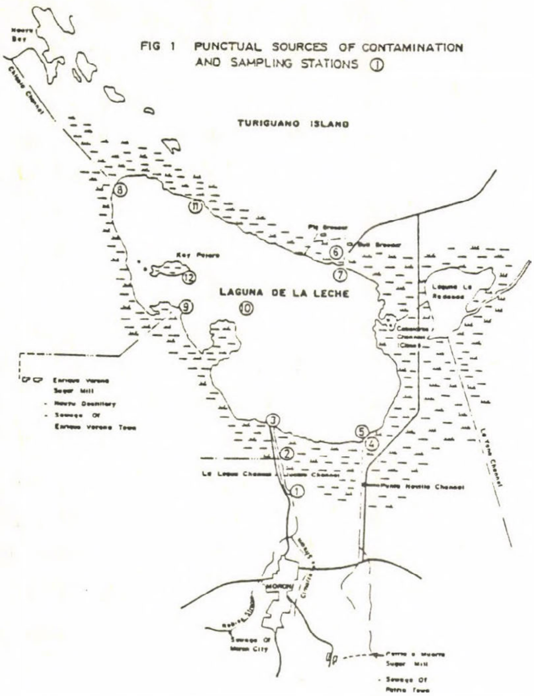


FIG 1 PUNCTUAL SOURCES OF CONTAMINATION AND SAMPLING STATIONS ①

TABLE I. MORPHOMETRIC FEATURES OF LAGUNA DE LA LECHE

SURFACE AREA AT FULL CAPACITY	66,15 km^2
MAXIMUM LENGTH	14,4 km
MAXIMUM WIDTH	7,5 km
SHORE LINE AT FULL CAPACITY	47 km
MEAN DEPTH	1,5 m
VOLUME	95,3 x 10^6 m^3

Results and discussion

A decrease in dissolved oxygen and pH as well as a rise of BOD and COD were observed at each station 30 days after the passage of the hurricane (Fig. 2). This is explained by enhanced decomposition of the organic matter accumulated in the lagoon due to the erosion of the basin. As a consequence of the intense phytoplankton production, concentration of dissolved oxygen and pH showed a further increase by the 125th day after the passage of the hurricane. At the same time BOD and COD values decreased and they were close to those observed during the first sampling.

High rates of decomposition resulted in a rapid regeneration of inorganic nutrients, particularly phosphate and ammonia (Fig. 3) as it could be observed on the 30th day after the cyclone. By 24 March 1986 concentration of both phosphate and ammonia decreased at each station indicating their in-

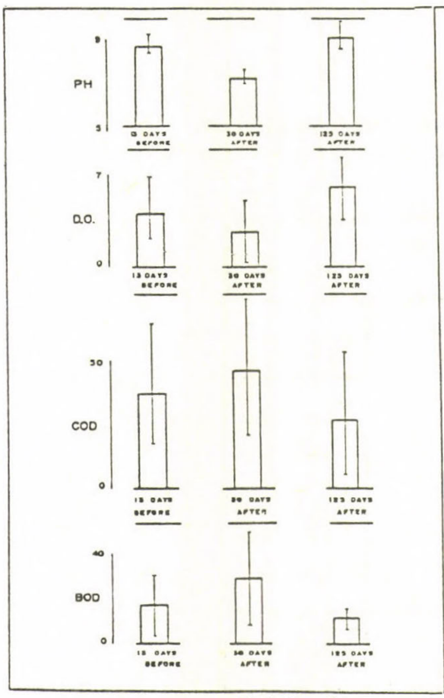


FIGURE 2

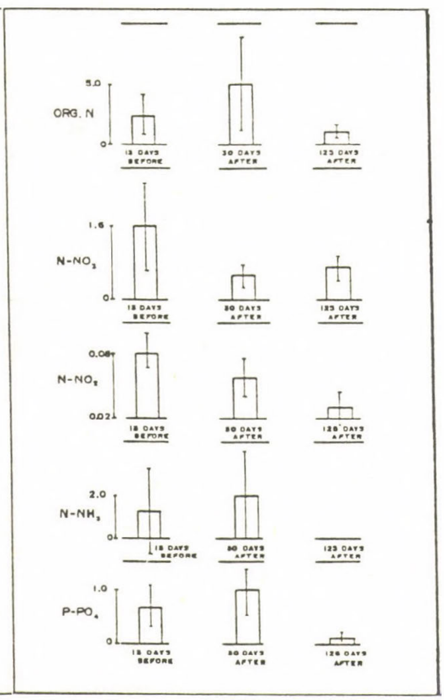


FIGURE 3

tense utilisation by the phytoplankton. It is well known that ammonia is the form of inorganic nitrogen preferentially utilised by most algal species (15).

Decreasing concentration of nitrite indicates that the improved oxygen conditions resulted in an enhancement of the nitrification in the lagoon.

In spite of the intense algal production an excess of nitrate could be observed. This is in agreement with the preferential utilisation of ammonia as a source of nitrogen.

Conductivity (Fig. 4) and alkalinity (Fig. 5) showed a decrease at each station 30 days after the cyclone in comparison to the first sampling due to the large volume of precipitation. On the last sampling their values were higher than during the first one. This can be related to the period of sugar-cane harvest as well as the industrial waste-water pollution reaching the lagoon by this time.

The dilution could also be verified by the analysis of the chloride content 30 days after the cyclone (Fig. 6). The increase observed 125 days after the cyclone was induced by convective currents mixing the bottom water of a high chloride content with more diluted surface water.

The number of total and fecal coliforms showed similar changes (Fig. 7). Due to the water renewal caused by the significant precipitation a marked decrease could be observed 30 days after the hurricane. The values, however, increased by the 125th day at each station, since large volume of sewage reached the lagoon at that time.

The number of phyto- and zooplankton was dramatically reduced by the cyclone with values close to zero. The explanation is that they were driven out from their biotopes and removed from the lagoon in large quantities through the Chicola Channel. The number of phyto- and zooplankton has increased by the last sampling.

Conclusions

- The passage of the hurricane caused an improvement in the water quality of the lagoon in both physico-chemical and

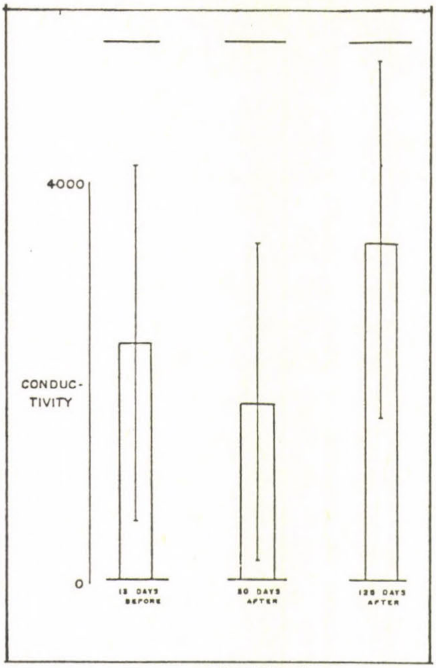


FIGURE 4

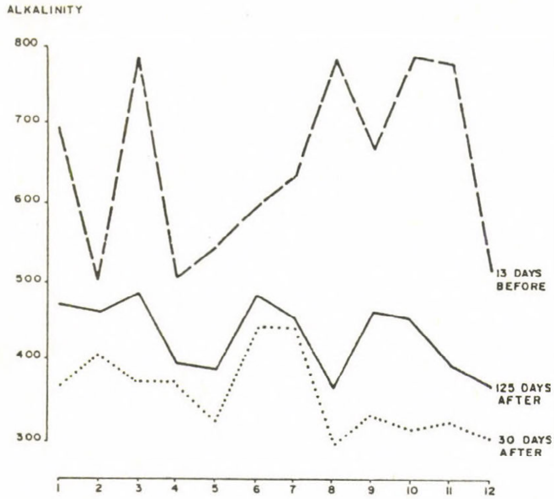


FIG. 5 ALKALINITY

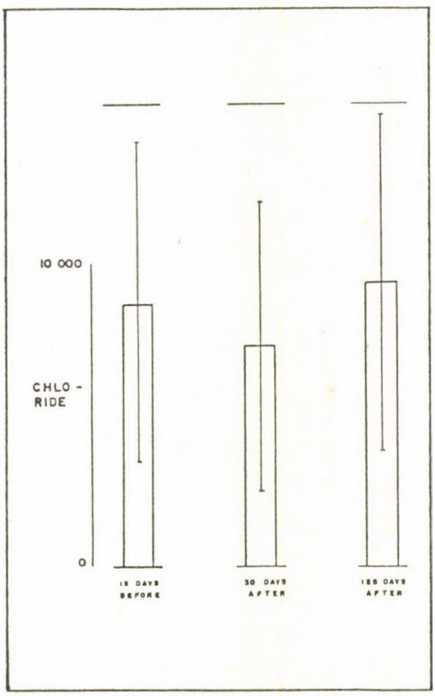


FIGURE 6

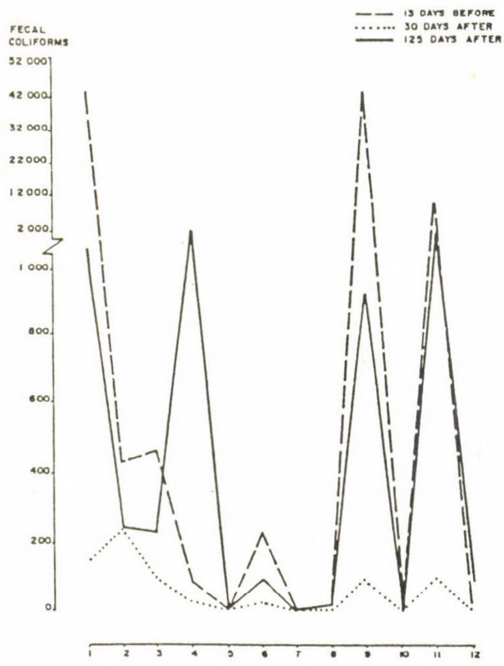


FIG. 7 FECAL COLIFORMS

biological terms, as it was indicated by the decrease of the number of coliform organisms as well as by the increased phyto- and zooplankton production.

- Enhanced decomposition of organic material resulted in a rapid regeneration of inorganic nutrients, such as ammonia and phosphate. This, in turn, led to enhanced algal production, the consequence of which was an increase in dissolved oxygen and pH and a decrease in the BOD and COD values.

- Large volume of precipitation significantly diluted the water of the lagoon.

References

1. Pingree, R.D., Holligan, P.M. and Mardelle, G.T. (1979). Phytoplankton growth and cyclonic eddies. Nature (Lond.), 278 (5701), 245-247.

2. Cintrón, G. and Schaeffer-Novelli, Y. (1981). Introducción a la ecologia del manglar. Seminario sobre la ordenación y desarrollo integral de las zonas costaneras. Guayaquil, Ecuador, 20 pp. EC 80/006.

3. Zeeman, S.I. (1985). The effects of tropical storm Dennis on coastal phytoplankton. Science, 20, 403-418.

4. Woodley, J.D. (1980). Hurricane Allen destroys Jamaican coral reefs. Nature, 287, 387.

5. Woodley, J.D. (1981). Hurricane's Allen impact on Jamaican coral reefs. Science, 214, 749-755.

6. Porter, J.W., Woodley, J.D., Smith, G.H. and Neigel, J.E. (1981). Population trends among Jamaican reef corals. Nature, 294, 249-250.

7. Knowlton, N., Lang, J.C., Rooney, M.C. and Clifford, P. (1981). Evidence for the layed mortality in Hurricane Damaged Jamaican staghorn corals. Nature, 294, 251-252.

8. Perez Eiriz, M.C. and Morales, T. (1985). Alteraciones producidas en las condiciones limnológicas de cuatro ecosistemas acuáticos en Cuba por efecto de ciclones. Resúmenes del Evento "A 20 años del Ciclón Flora". Holguín.

9. Verma, M.H. (1969). Hydrobiological study of a tropical impoundment Tekampur Reservoir, Gwalier, India, with special reference to the breeding of Indian carps. Hydrobiologia, 34, (3/4), 358-368.
10. Ganapati, S.V. (1973). Ecological problems of man-made lakes of South India. Arch. Hydrobiol. 71, 363-380.
11. Battoe, L.E. (1985). Changes in vertical phytoplankton distribution in response to natural disturbances in a temperate and a subtropical lake. J.Fresh.Ecol. 3 (2), 167-174.
12. APHA - AWWA - WPCF (1985). Standard methods for the examination of water and waste waters. Washington D.C., 16th ed.
13. Javornicky, P. (1958). The revision of some quantitative methods for phytoplankton research. Sci.Pap.Inst.Chem.Technol. Fuel and Water, 2 (1), 283-367.
14. Straškraba, M. and Hrbacek, J. (1966). Net plankton cycle in Slapy reservoir during 1958-1960. Hydrobiol.Stud. 1, 113-153.
15. Muhlhauser, H. (1984). Nutrientes y productividad primaria. In: Embalses Fotosintesis Productividad Primaria. Bahamon de N. y Cabrera S. (eds). Programa sobre el Hombre y la Biosfera. Unesco, pp. 149-154.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

EUTROPHICATION OF LAKE PARANOÁ AND DEVELOPMENT OF BRASILIA

L. SOMLYÓDY,* I.A. MASINI,** G. van STRATEN***

*Research Center for Water Resources Development, VITUKI
Budapest, Hungary
**Brasilia Water and Sewage Corporation, CAESB
Brasilia, Brazil
***Twente University of Technology
Enschede, The Netherlands

1. INTRODUCTION

Lago Paranoá, an artificial lake of 38 km^2 surface area and 510.10^6 m^3 volume, was created in 1961 for forming the major recreational center of Brasilia, the new capital of Brazil (Fig.1). Unfortunately, the first sewage treatment plants started to operate in 1966, only and the lake is the recipient of untreated or inadequatly treated waste waters even today, since population growth has been much faster than expected. Due to increasing nutrient loads the lake water became hypertrophic and unsuitable for body contact recreation already by the mid seventies. In 1978 a striking bloom of Microcystis aeruginosa was observed (biomass values exceeded 500 mg/l) leading to the preparation of an emergency program based on the application of copper sulphate. Since that time algicide is being applied regularly as a tool for treating the symptoms rather than the causes of eutrophication.

The process of nutrient enrichment was studied in detail in the frame of a UNDP/WHO project between 1975 and 1977 (Björk, 1979) and a comprehensive list of measures was defined for restoring the lake. This project helped considerable also in respect of establishing a monitoring network covering the lake, its tributaries and sewage discharges.

At the early eighties a recovery program was worked out by CAESB focusing on sewage treatment. Two advanced treatment plants with biological phosphorus- and nitrogen removal was proposed to construct in the vicinity of those in operation (Fig.1). The total capacity is 210000 m^3/d corresponding to 700000 inhabitants forecasted as the saturation value for Brasilia (Fig.1). Due to financial difficulties the construction has been started in 1987, only.

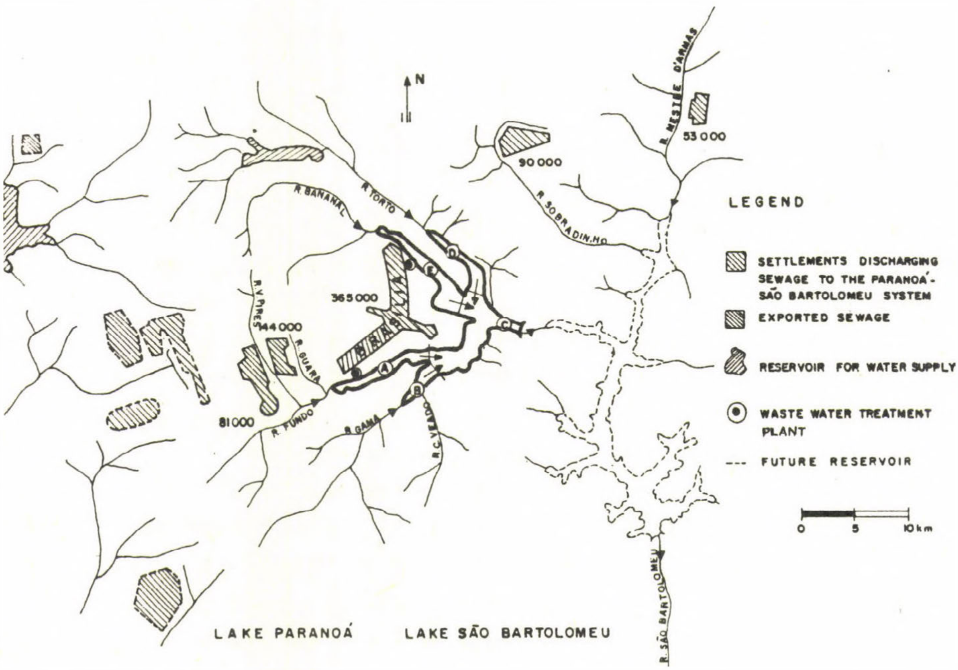


Fig.1 Lake Paranoá and the region of the Federal District

In the meantime, the total population of the Federal District including Brasilia, several satellite cities (Fig.1) and other settlements exceeded 1700000, the drinking water demand of which can be just met utilizing the two existing reservoirs (see Fig.1). As population estimates for 2015 range between

3000000 and 6000000, additional resources should be looked for. One of the major alternatives is the construction of a new reservoir called Lago Sao Bartolomeu (Fig.1) which (depending on its size) will be fed by the outflow of Lake Paranoá and consequently its quality is a function of the trophic state of the existing lake. Thus, a complex, regional eutrophication and water resources management task is given, which is studied since 1987 in the frame of a comprehensive program supported by UNDP.

The approach adopted is based on the principle of decomposition and aggregation developed for analysing the eutrophication management problem of Lake Balaton (Somlyódy, 1982, Somlyódy and van Straten, 1986). Major elements together with their linkages are illustrated in Fig.2 showing clearly the impact of

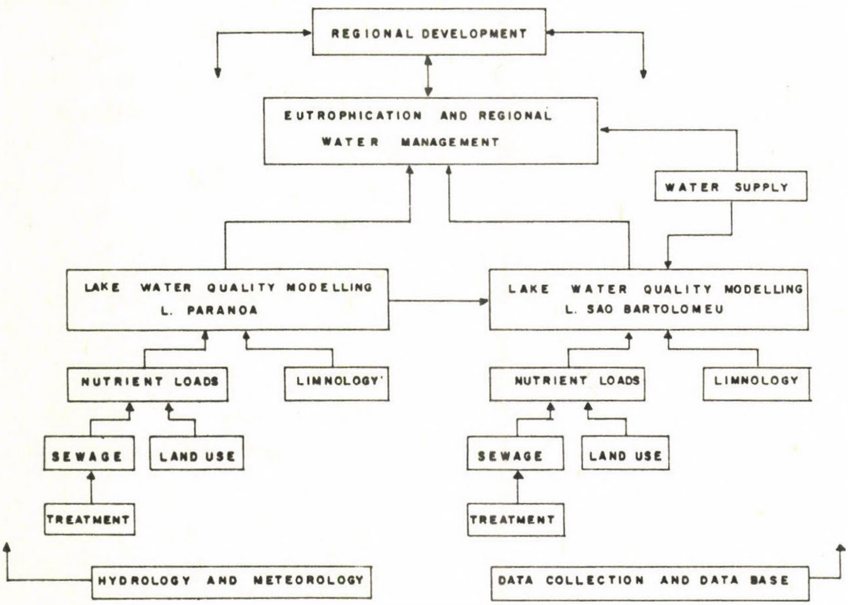


Fig.2 Application of the principle of decomposition and aggregation to the problem of Paranoá - Sao Bartolomeu Basin

the Paranoá basin (to be controlled) on the water quality of the reservoir planned and the associated water treatment technology.

Subsequently we summarize some of the results achieved until now in the descriptive phase of the study, primarily in relation to nutrient loads and phosphorus household of Lake Paranoá.

2. NUTRIENT LOADS

More than 600 000 people live at present in the watershed of Lake Paranoá, which basically determine phosphorus and nitrogen loads of the lake. Approximately half of the sewage is diverted to the two treatment plants (Fig.1). Here, about 80 % is processed, while the rest is directly discharged to Lake Paranoá (by-pass). The other half of waste waters is treated partially, only, and reaches the lake mostly via the Fundo River (see Fig.1).

Earlier total phosphorus (TP) load estimates prepared on the basis of data of the monitoring network resulted in values between 208 kg/d and 281 kg/d (Somlyódy, 1987), which range is not in harmony with the number of inhabitants of the basin.

For exploring this discrepancy, several studies were performed as follows:

(i) longitudinal water quality profile measurements were made in the subwatershed of River Fundo (and also in the basin of River Sao Bartolomeu) under wet and dry weather conditions;

(ii) daily observations were executed at the mouth section of River Fundo;

(iii) diurnal fluctuations were studied, too, in three cross-sections at the junction of Rivers C. Guará and Fundo (Fig.1);

(iv) the fate of sludge of the two treatment plants was monitored;

(v) measurements and estimates were made on urban runoff.

From the above experiments the subsequent conclusions could be drawn:

(i) The load carried by River Fundo is primarily influenced by waste waters discharged to the river, and thus the relation of TP concentration to flow rate (Q) shows a hyperbolic character, in the downstream sections.

(ii) Each load component is subject to a pronounced diurnal variation at the mouth reflecting the superimposed effect of six major upstream discharges at the distance of the travelling time ranging between 1 and 4 hours. TP load is small in the morning, doubles around lunch time and attenuates only at 8-10 p.m. Since in the frame of the regular monitoring program samples are taken in morning hours, the load values obtained are underestimated by about 50 %. This finding is valid both for dry and wet seasons.

(iii) Phosphorus retention by rivers was found very effective (dry season). This statement applies not only for TP, but also for PO_4-P and particulate phosphorus (PP), as can be seen from Fig.3. showing the impact of the city of Planaltina on the quality of River Mestre D'Armas ($Q \cong 1$ m^3/s, Sao Bartolomeu basin, see Fig. 1). Assuming first order kinetics for the removal (governed by sedimentation and probably by adsorption) a rate of about 12 d^{-1} was obtained for each component. Average retention was slightly above 60 % (~10 %/km). For R.C. Guará ($Q \cong 0.5$ m^3/s), the tributary of River Fundo approximately 50 % retention was found.

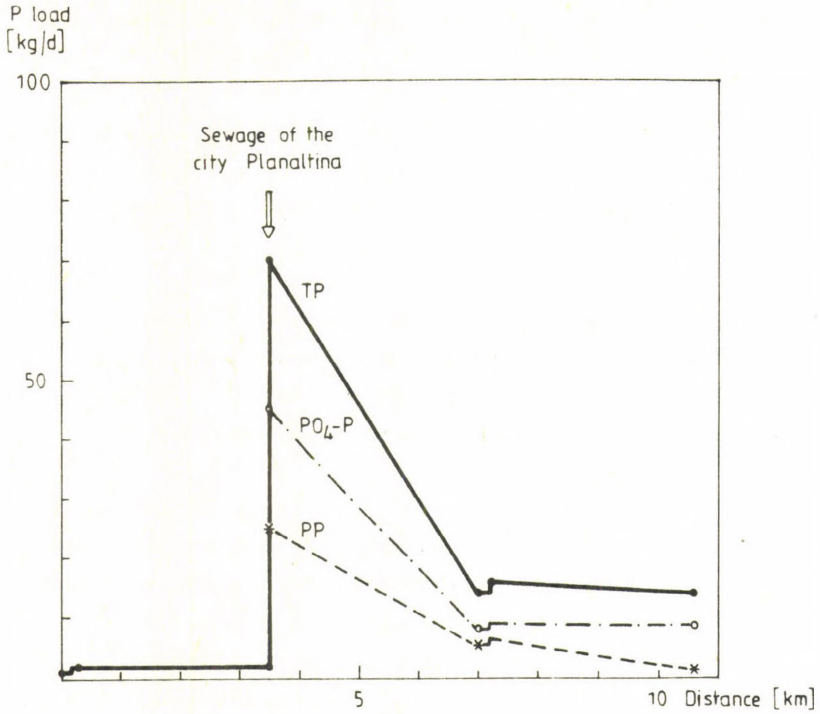


Fig. 3 TP retention by the River Mestre D'Armas

(iv) From the data of treatment plants 1.6 gP/pe.d and 270-350 lit/pe.d values could be derived for P release and sewage discharge, resp. (pe: population equivalent)

(v) The total P removal efficiency at the two treatment plants does not exceed 20-30 %, due to the inadequate sludge handling and disposal capacity.

(vi) The first observations on urban runoff suggest a unit areal load of about 0.2 kgP/ha.y (which is probably an underestimate as the rainfall intensity was relatively low).

(vii) A stratified sampling program was introduced on the basis of analysing the data of the daily observations.

Accordingly, frequency is one per month for dry and 2-4 per month for wet season depending on the importance of cross section.

The studies outlined above revealed that the annual average TP load of the lake is 713 kg/d ($\sim 18.5\ mg/m^2d$) for the period from July 1986 to June 1987, well representing P sources in the watershed, retention processes and the degree of sewage treatment (this also provides a solid basis to prepare a pollution inventory for the basin). Details for various load components and segments of the lake (see Fig. 1) are given in Table 1. It is striking to observe the unit areal load of Branch A which is 22 times higher than in the middle part of the water body (Branch C).

It is noted that although the ratio of TN to TP loads is slightly below 10, still however phosphorus is the key element of controlling eutrophication of Lake Paranoá as the dominating blue-green algae becomes self-supporting with atmospheric nitrogen when nitrogen is depleted in the water (Björk, 1979).

3. PHOSPHORUS BALANCE OF LAKE PARANOÁ

Before starting to discuss details of the phosphorus household a brief evaluation is given on major limnological properties of the lake.

The temperature of Lake Paranoá ranges between 20 and 27 ^{o}C throughout the year. Stratification occurs only in Branch C (Fig.1) which stopps in June-July when the single overturn induces strong mixing in the entire water body. Due to the geological conditions in the catchment area the concentration of ions is extraordinary low. The pH fluctuates between 7 and 10 in the epilimnion. In the hypolimnion total lack of oxygen is measured except for the period of overturn. Results of water quality observations are in harmony with the load pattern of Table 1, which also shows the strong spatial variability of TP.

Seasonal changes are much smaller than for lakes in the temperate zone, as light and temperature conditions are favourable all the time. The plankton community can be characterized as a monoculture of Anabenopsis raciborskii, however, Mycrocystis blooms make the most striking impression for the public. Algal biomass and Chl-a concentration can not be used for characterizing eutrophication because of the regular dosage of copper sulphate (see earlier). For this reason TP is the best suited and simple trophic indicator.

Phosphorus cycling is very fast in the lake and zooplankton does not seem to play an essential role. PO_4-P concentration is low all the time in the open water because of algal uptake. Probably the fast mineralization is the explanation why P accumulation can not really be observed in the hypolimnion and the sediment is poorer in P than expected from results of mass balance computations on phosphorus and suspended solids. This also suggests that the recovery time after a load reduction is much shorter than for a temperate lake (see also Björk, 1979).

Subsequently, our attention is turned to TP balance computations. First, steady state conditions and five completely mixed reactors for Branches A...E as indicated in Fig. 1 are assumed. Our objective is to evaluate TP retention of various segments and to compare apparent settling rates (v_s) obtained from the Vollenweider type of relatonship (Vollenweider and Kerekes, 1982) and actual in-lake observations, resp.

For the first the expression

$$v_s = q_s \quad T_w^{1/2} \qquad (1)$$

holds, while for the second (neglecting the covariance term)

$$v_s = q_s \left(\frac{1}{P} \ \frac{l_s}{q_s} \ - 1 \right), \qquad (2)$$

where notation is given in Table 1. and all the values represent annual averages. Details of the calculations together with major geometric and hydrologic data are presented in Table 1. The evaluation of results leads to conclusions as follows:

(i) The Vollenweider approach leads to settling velocities smaller by an order of magnitude than those obtained from observations through mass balance calculations. This means that the TP retention of the lake is much larger than predicted by Eq(1).

(ii) Settling velocities are very high in Branches A and E, which are the most polluted segments of the lake. Phosphorus retention is extremely effective in both segments, 92 % and 94 %, resp. Those represent 80 % from the total P removal of 96 % in the lake.

In the second step of the analysis a dynamic TP balance model operating on a monthly basis has been developed (van Straten, 1987) assuming also a first order net removal process and the same segments as before (to which the hypolimnion of Branch C was added). Calibration was made for 1983 and 1984, while the rest of the period 1978-1985 was utilized for validation (Fig.4). For apparent settling rates slightly lower values have been obtained than indicated in Table 1 (e.g. 0.50 m/d and 0.12 m/d for Branches A and E, resp.), however loads employed were also underestimated. Model simulation for Branch A shows relatively poor agreement with observations. The situation has not been improved when the plug flow character of side-arms has been better approximated by segmenting the branch into tanks-in-series. There was however some experimental evidence that P removal takes place very fast, within a couple of hundred meters downstream of external loads. For this reason an immediate removal rate subject to calibration was introduced on the loads of River Fundo and the South treatment plant (Fig.1), resp. This hypotheses led to considerable model improvement (here Branch A was subdivided into two segments), as can be

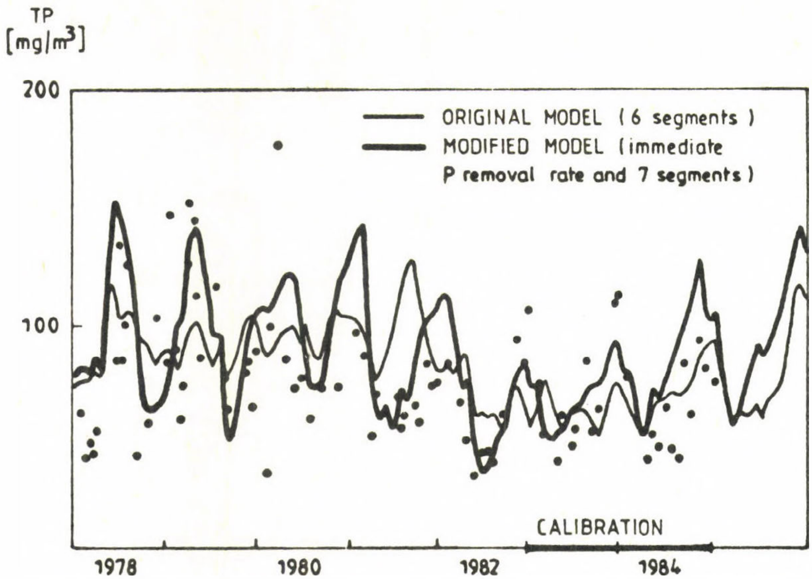


Fig. 4 Results of dynamic TP balance models for Branch A

judged from Fig. 4. Parameter estimation has shown that 80-85 % of TP is settled immediately in the water body which actually could not be described properly when employing the usual kinetics for sedimentation (not even if the power of concentration was increased).

The hypotheses of the immediate P reduction is well supported by Fig. 5 showing average values of three subsequent longitudinal profile observations performed in Branch A. As can be seen P removal takes place within short segments downstream of the discharge points. It is worthy to note that average retention of Branch A (the TP load of which was above 500 kg/d, see Table 1 for the annual average) exceeded 95 % for the week of the experiment, which is in good agreement with high removal rates found through mass balance computations.

Table 1. Settling velocities obtained by the Vollenweider approach and computed from mass balance (July 1986 - June 1987). (*denotes throughflow from other branches)

BRANCH		A	B	C	D	E	TOTAL
Volume (V)	(10^6m^3)	39.34	31.00	256.57	65.42	117.27	510.00
Outflow (Q_{out})	(m^3s^{-1})	4.11	1.82	10.23	1.33	2.16	10.23
Filling time ($T_W=\frac{V}{Q_{out}}$)	(y)	0.31	0.54	0.80	1.56	1.72	1.58
Area (A)	(10^6m^2)	4.59	3.08	15.26	5.82	9.73	38.48
Hydraulic load(q_s)	(my^{-1})	28.24	18.88	21.14	7.21	7.00	8.38
Settling velocity (Eq.1)	(md^{-1})	0.04	0.04	0.05	0.02	0.03	0.04
Load components	(kg d^{-1})						
Sewage		231.12				114.17	345.29
By-pass		37.30				45.40	82.70
Tributaries		154.50	11.13	54.42*	5.68	3.86	175.17
Direct vicinity		24.44	1.88	15.98	1.88	2.82	47.00
Stormwater		1.77	0.41	2.05	1.64	1.02	6.89
Precipitation		0.27	0.15	0.76	0.53	0.29	2.00
Total load (L)	(kgd^{-1})	449.40	13.57	73.21	9.73	167.56	713.47
Unit areal load ($l_s=\frac{L}{A}$)	($10^3mgm^{-2}y^{-1}$)	35.73	1.61	1.75	0.61	6.29	6.77
TP concentration (1m depth) (P)	(mg m^{-3})	107.70	24.70	34.60	22.80	52.17	34.60
Settling velocity from mass balance (Eq.2)	(md^{-1})	0.83	0.13	0.08	0.05	0.31	0.51
Load at the outflow	(kg d^{-1})	38.10	3.88	30.58	2.62	9.73	30.58
Retention	(%)	91.52	71.41	58.23	73.07	94.19	95.71

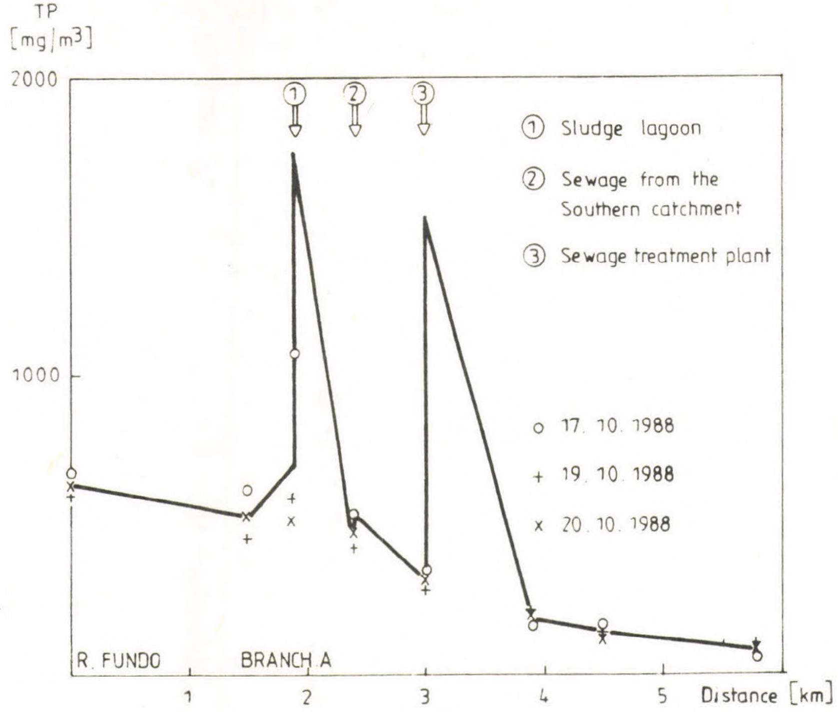


Fig.5 Longitudinal profile observations in Branch A (1 m depth)

For analysing details of fast P reduction in Branch A a model incorporating four fractions (dissolved-, algal particulate-, non-algal particulate- and particulate inorganic P) is under development (van Straten, 1988, Kouwenhoven, 1988).

4. FUTURE DEVELOPMENTS

In subsequent steps of the study the water quality of Lake Paranoá will be evaluated for various control alternatives (as mentioned in the Introduction two advanced sewage treatment plants are under construction). Similar studies and model developments as outlined in the paper are being performed for the system of Sao Bartolomeu. Here the objective is to evaluate the water quality of the future reservoir for different possible sizes (as a function of water demand and hydrologic regime) together with its dependance on the trophic state of Lake Paranoá. The latter, in turn, is determined by regional development, water resources management and pollution control in the Paranoá basin. No formal optimization is planned to handle the above complex problem, but a set of scenarios and alternatives will be analysed by a coupled planning type watershed nutrient load model and a P balance model in an off-line fashion.

SUMMARY

Together with the construction of Brasilia (the capital of Brazil) a tropical reservoir called Lake Paranoá aimed at serving recreational purposes was also created. Unfortunately, however, the water body has reached a hypertrophic state within a short period of time due to partial and inadequate sewage treatment and intensive population growth. The latter is going to continue and the exploration of new resources for water supply is necessitated. The major alternative is to build a second reservoir, which however may be fed by the hypertrophic Lake Paranoá depending on its size . Thus, a complex, regional eutrophication and water resources task is given which is studied in the frame of a project supported by UNDP.

This paper offers a summary of results achieved until now mainly in relation to Lake Paranoá. A detailed estimate is given on TP load which on the basis of comprehensive field investigations turned out to be three times higher than earlier figures. The value obtained corresponds well with the number of inhabitants of the region, per capita P release values, retention processes and degree of sewage treatment. TP retention was found fairly high in rivers (50-60 %). Steady state and dynamic TP balance computations performed for the lake have indicated apparent settling rates by order of magnitude larger for side-arms (0.3-0.8 m/d) than estimated by the Vollenweider relation. Phosphorus retention of the lake is

about 95 %, 80 % of which takes place in the heavily loaded two, riverine type segments of the water body. As justified also by in-lake observations, here P removal occures within a couple of hundred meters long reach downstream of discharges which could not have been simulated by assuming the usual kinetics for sedimentation. The introduction of an immediate P removal rate led, however, to considerable improvement in model performance. The explanation of the above net mechanism is further studied by a model incorporating four P fractions.

Subsequently models for both water bodies, together with planning type nutrient load models will be used in the frame of a scenario analysis to consider the regional water resources management problem of the entire region.

ACKNOWLEDGEMENTS

We sincerely acknowledge the contribution of our colleagues at the Ambiental Technology Directorate of CAESB to experimental, laboratory and other research works of the study, the results of which were utilized here.

REFERENCES

Björk, S. (1979) The Lago Paranoá Restoration Project. Technical Report (manuscript)

Kouwenhoven, P. (1988) Assessment of Sanitary and Environmental Characteristics of the Sao Bartolomeu River and Lake Paranoá Basins: Water Quality Modelling (UNDP Project BRA 87/011/B/01/01). Report on a consultancy visit (manuscript)

Somlyódy, L. (1982) Modelling a complex environmental system: The Lake Balaton Study."Mathematical Modelling" 3, pp. 481-502.

Somlyódy, L. and van Straten, G. (eds) (1986) Modelling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton. Springer Verlag. Berlin

Somlyódy, L. (1987) Assessment of Sanitary and Environmental Characteristics of the Sao Bartolomeu River and Lake Paranoá Basins (UNDP Project BRA/87/011/B/01/01) Report on a consultancy visit (manuscript)

van Straten, G. (1987, 1988) Assessment of Sanitary and Environmental Characteristics of the Sao Bartolomeu River and Lake Paranoá Basins (UNDP Project BRA/87/011/B/01/01) Reports on consultancy visits (manuscript)

Vollenweider, R. A. and Kerekes, J.J. (1982) Background and Summary Results of the OECD Cooperative Programme on Eutrophication. OECD Report (Paris)

Symp.Biol.Hung. 38 (1989) *Conservation and Management of Lakes*
Akadémiai Kiadó, Budapest *J. Salánki, S. Herodek (eds)*

SOME ASPECTS OF THE IMPACT OF TOURISM ON MONDSEE, AUSTRIA

M. DOKULIL, A. JAGSCH*

Institut für Limnologie der Österreichischen Akademie der Wissenschaften, A-5310 Mondsee, Gaisberg 116, Austria
*Bundesanstalt für Fischereiwirtschaft, A-5310 Scharfling, Austria

1. INTRODUCTION

Tourism can influence lake ecosystems in many ways both directly and indirectly. One of the major impacts is through eutrophication resulting from increased nutrient input to the system originating from large touristic populations. Such impacts have long been recognized (Vollenweider 1968) and were extensively analysed in recent years (e.g. Uhlmann Ed. 1976; OECD 1980a,b,c; Loehr et al. eds. 1980). Phosphorus has commonly been identified as the main nutrient controlling stagnant freshwaters.

Rehabilitation and restoration concepts therefore usually include complete sewage diversion from the system or phosphorus precipitation in treatment plants (e.g.Dunst et al. 1974; Stumm ed. 1987).

Other causes come from recreational activities such as swimming, boating and angling. More indirect influences such as the increase in salt content of waters due to de-icing of roads have recently attained some interest (Sampl 1980, 1988).

In the present study we report on the time course of eutrophication and rehabilitation in a deep, alpine lake in Austria. In addition we give evidence for increased chloride content resulting from the application of de-icing agents.

2. THE SYSTEM

Mondsee is a moderately large lake, surface area 14.2 km^2, with a maximum depth of 68.3m situated in the "Salzkammergut" lake region of Austria (Fig. 1). Mondsee belongs to a chain of lakes receiving affluents from two other water bodies and draining through its 2.6 km long outflow into Attersee, Austria's largest lake. The theoretical water renewal time is ca. 1.7 years. The catchment area (247.2 km^2) largely comprises agricultural land and forests. Excessed by one of Austrias main highways, Mondsee became the touristic center of the region.

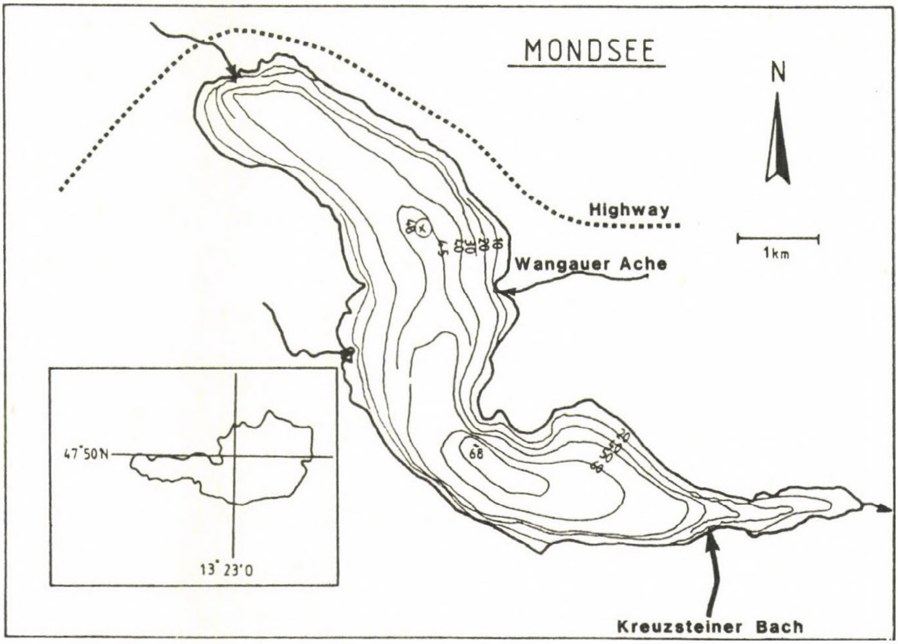


Fig. 1. Location and contour map of Mondsee, Austria

3. EUTROPHICATION

The rapid development of tourism and recreation in the mid-fifties, expressed here as the number of overnight stays (Fig. 2) led to first signs of responds of the lake to the elevated nutrient concentrations due to enhanced sewage input in 1968 (Danecker 1969; Findenegg 1969), when Oscillatoria rubescens D.C. was first noticed (Fig. 3). The decade from 1957 to 1967 was characterized by relatively low phytoplankton biomasses which were greatly depressed during the early sixties (Fig. 3) due to large amounts of inorganic material washed into the lake during the construction of the highway on the northern slopes (Einsele 1963).

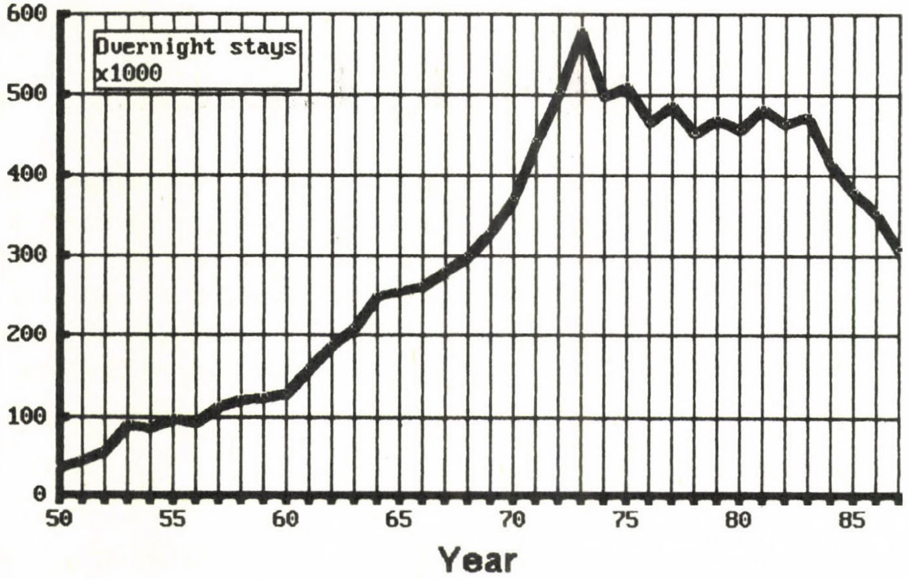


Fig. 2. Development of tourism around Mondsee expressed as overnight stays in the years 1950 to 1987.

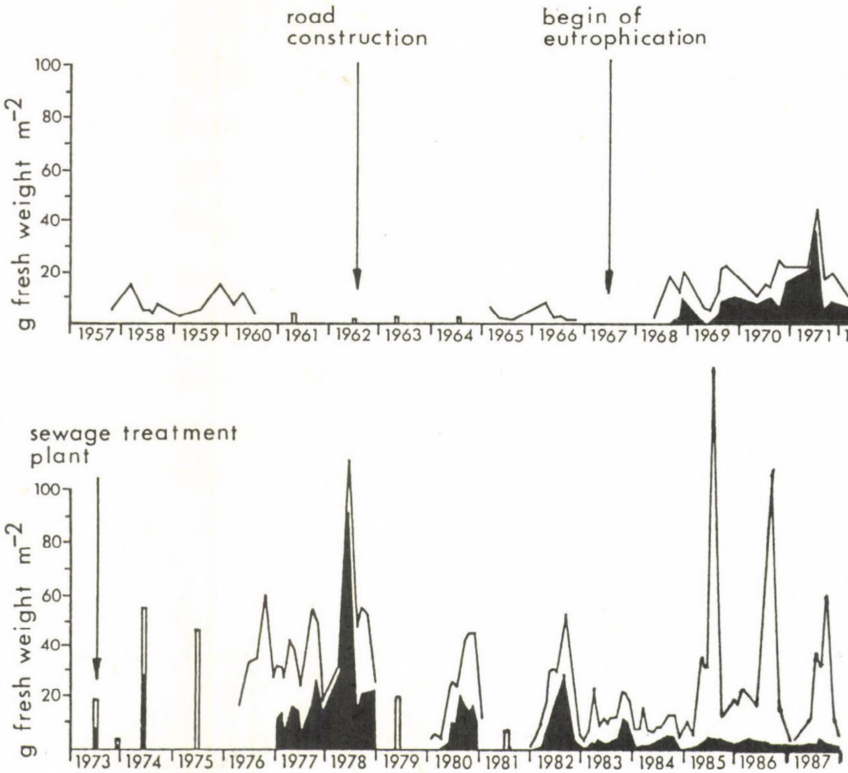


Fig. 3. Long term development of algal biomass in Mondsee (0 - 20 m) for the years 1957 - 1987. The shaded area is the contribution of <u>Oscillatoria</u> <u>rubescens</u>. Data are from Findenegg (1957 - 1972), Müller-Jantsch (1975, 1976), Oberrosler (1977), Schwarz (1978, 1980), Moog (1979, 1981) and Dokulil (1982 - 1987). For the years 1973 and 1974 old, uncounted samples from Findenegg were analysed.

Corresponding with the further development of tourism (Fig. 2) algal biomass and the <u>Oscillatoria</u> contribution therein increased culminating in massive <u>Oscillatoria</u> blooms in the early seventies (Fig. 3). Largest phytoplankton biomass was reached in 1978 declining thereafter.

Due to the high phytoplankton biomass oxygen over-saturation up to 190% occured in the epilimnion and oxygen depletion in the hypolimnion during the summer stagnation (Jagsch 1979). Total phosphorus content of the lake during spring overturn was high during the seventies peaking in 1979 (Fig. 5).

This course of the eutrophication process is clearly documented in the fossil records of the sediment. Deep sediment layers are free of Oscilloxanthin, a carotenoid characteristic for Oscillatoria. Concentrations therafter increase during the eutrophic phase (Schultze 1985). Similar changes are reflected in the diatom flora of the lake. Tabellaria flocculosa var. asterionelloides among others became dominant in the laminated sediments typical for the eutrophic phase (Schmidt et al. 1985; Klee & Schmidt 1987). These sediments contain high amounts of phosphorus (Helbig et al. 1985). Dramatic changes also took place in the Ostracod populations during eutrophication (Danielopol et al. 1985).

4. REHABILITATION

For the restoration of the lake a sewage diversion system was planned and the treatment plant started to operate in 1973. Since complete diversion of the treated effluents from the catchment was not possible because of the short 2.6 km long outflow draining into another lake, phosphorus precipitation was incorporated into the sewage plant and the treated effluents discharged into the lake.

As can be seen from Fig. 3 no immediate response of the system was observed. Instead algal biomass peaked in 1978 and then started to decline partly as a result of climatic conditions

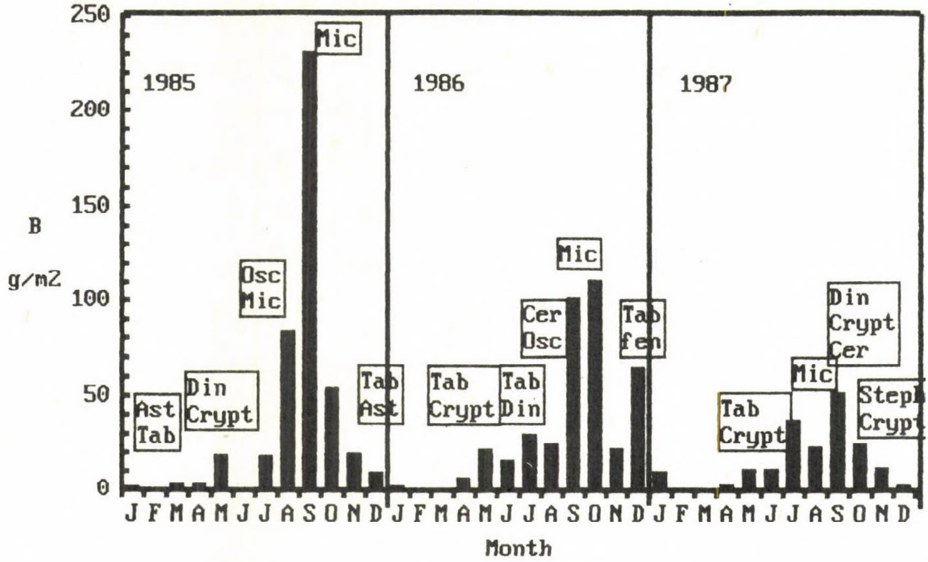


Fig. 4. Phytoplankton biomass in the euphotic zone as g m^{-2} fresh-weight for the years 1985 to 1987. Ast = Asterionella formosa; Tab = Tabellaria flocculosa var. fenestrata; Din = Dinobryon divergens; Crypt = Cryptomonas spp.; Osc = Oscillatoria rubescens; Mic = Microcystis aeruginosa; Cer = Ceratium hirundinella; Steph = Stephanodiscus spp..

(high precipitation in 1979). Investigations by Schwarz (1979, 1980) demonstrated a considerable reduction in Oscillatoria biomass which has further declined since (Dokulil & Skolaut 1986; Dokulil 1987). This was considered as a sign of reoligotrophication (Dokulil 1984) due to raising numbers of sewer connections and better treatment. These changes are also reflected in the diatom population where Melosira islandica replaced M. italica ssp. subarctica, while Tabellaria became less important (Klee & Schmidt 1987).

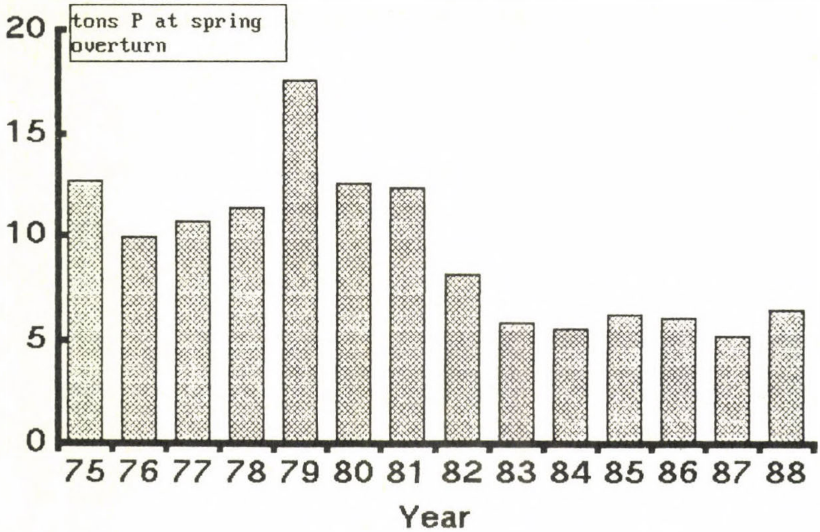


Fig. 5. Total phosphorus content in tons at spring overturn for Mondsee for the years 1975 to 1988.

Recently, since 1985, Microcystis aeruginosa forms large maxima in autumn (Fig. 3 and 4) accompanied by other blue-green algal species including Aphanizomenon flos-aquae (Dokulil 1987). The maxima in fall decreased in the years 1986 and 1987 partly due to high wash-out. August 1988 was characterized by a large bloom of Dinobryon spp.. These events seem to indicate an unstable situation and changes taking place in the lakes environment.

During the recovery period total phosphorus content at spring overturn declined considerably between 1979 and 1983 and remained stable in the years after (Fig. 5).This coincides with the improvement of the sewage diversion from the lake and the declining numbers of overnight stays (Fig. 2).

According to Moog (1987) phosphorus loading declined from 1846 mg $m^{-2}a^{-1}$ in the year 1979 to 646 mg m^{-2} a^{-1} in 1984. This last figure still is 9% higher than the critical loading.
The trophic situation of Mondsee has accordingly improved from eutrophic to oligo-mesotrophic conditions.

5. CHLORIDE IMPACT

Heavy traffic partly due to tourism requires de-icing of the highway and several other main roads during winter. About 900 tons de-icing salt per year are applied to that part of the highway within the catchment area. Considerable amounts of the chloride contained in the de-icing agents are ultimately washed into the lake by the tributaries. As shown in Figure 6 the chloride content of the lake more than doubled in a period of 16 years (1972 - 1988). The strong impact through the de-icing of the highway is examplified by the high chloride concentrations from a tributary (Wangauer Ache) draining the area near the highway (Fig. 2). Distinct peaks appear during winter (Fig. 7). From the same figure it becomes clear that these concentrations are dependent from the de-icing of the roads since a tributary in the south (Kreuzsteinbach, Fig. 2) draining an uninhabited area displays natural background values of chloride. Compared to other areas in the region natural chloride concentrations in the lake were low originally (ca. 3 mg l^{-1}).
Similar observations were made in other lakes of Austria (Sampl 1980, 1988). Concentrations are not yet critical but further development should be kept under control because density

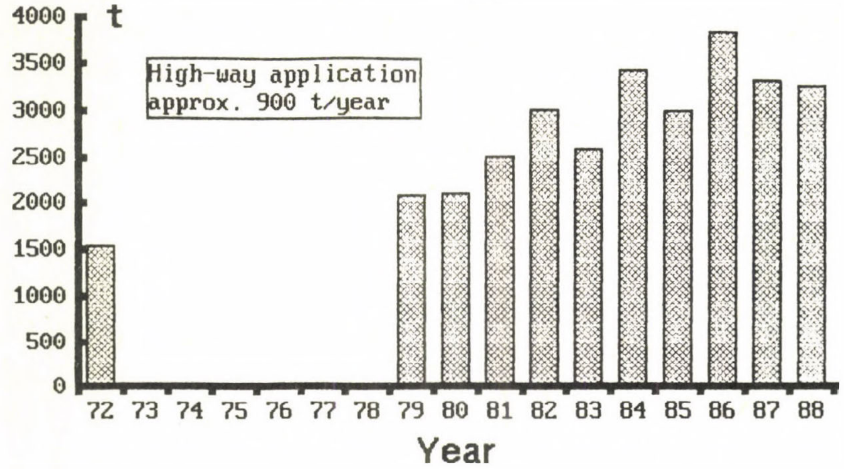


Fig. 6. Chloride content of Mondsee in tons.

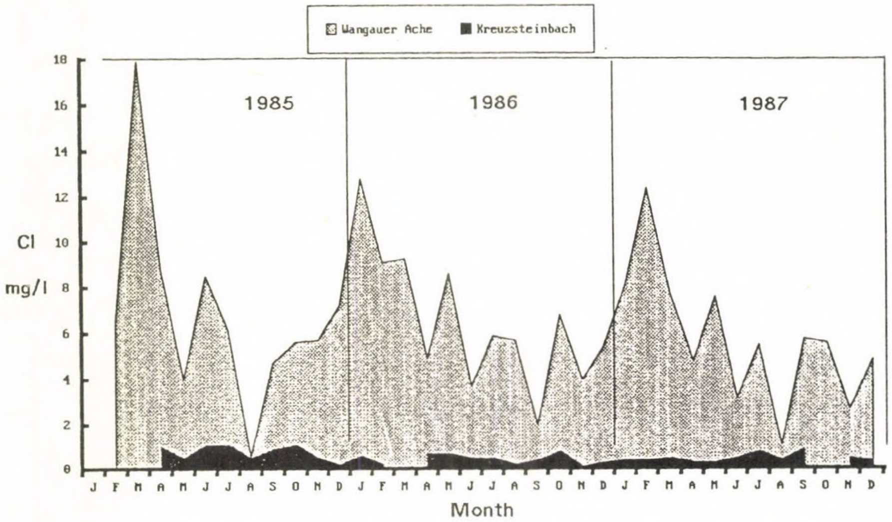


Fig. 7. Chloride concentrations of two tributaries of Mondsee in mg/l for the years 1985 - 1987.

dependent changes of mixing might occur as was found by Steinberg & Schrimpf (1982) in a Bavarian lake. In Mondsee changes might be delayed because of the high flushing rate and the wind exposure.

6. CONCLUSION

As has been shown impact by tourism can remarkably change the trophic status of a lake through increased nutrient input. Rehabilitation may take some time, in the present case about 10 years. Among other impacts, not mentioned here, indirect influences such as the increased chloride load discussed above may strongly affect lake environments.

7. REFERENCES

Danecker,E., 1969. Bedenklicher Zustand des Mondsees im Herbst 1968.- Österr. Fischerei. 22:25-31.

Danielopol,D., W. Geiger, M. Tölderer-Farmer, C.P. Orellana & M.N. Terrat, 1985. The Ostracoda of Mondsee: Spatial and temporal changes during the last fifty years. - In: Contributiones to the paleolimnology of the Trumer Lakes (Salzburg) and the lakes Mondsee, Attersee and Traunsee (Upper Austria). (Eds. D. Danielopol, R. Schmidt & E. Schultze),99-121, Limnologisches Institut, Österr. Akad. Wiss.

Dokulil,M., 1984. Die Reoligotrophierung des Mondsees. Laufener Seminarbeiträge ANL 2/84:46-53.

Dokulil,M., 1987. Long term occurence of blue-green algae in Mondsee during eutrophication and after nutrient reduction, with special reference to Oscillatoria rubescens. - Schweiz. Z.Hydrol. 49:378.

Dokulil,M. & C. Skolaut, 1986. Succession of phytoplankton in a deep stratifying lake: Mondsee, Austria.- Hydrobiologia 138: 9-24.

Dunst,R.C., S.M. Born, P.D. Uttomark, S.A. Smith, S.A. Nichols, J.O. Peterson, D.R. Knauer, S.L. Serns, D.R. Winter & T.L. Wirth, 1974. Survey of lake rehabilitation techniques and experiences. - Tech. Bull. 75:1-176. Dept. Natural Res., Madison, Wisconsin.

Einsele,E., 1963. Schwere Schädigungen der Fischerei und der biologischen Verhältnisse im Mondsee durch Einbringung von lehmig-tonigem Berg-Abraum. - Österr. Fischerei 16:2-12.

Findenegg,I., 1959. Das pflanzliche Plankton der Salzkammergutseen. - Österr. Fischerei 36:241-244.

Findenegg,I., 1969. Die Eutrophierung des Mondsees im Salzkammergut, - Wasser und Abwasser Forschung 4:139-144.

Findenegg,I., 1971. Die Produktionsleistung einiger planktischer Algenarten in ihrem natürlichen Milieu. - Arch. Hydrobiol. 69:273-293.

Findenegg,I., 1973. Vorkommen und biologisches Verhalten der Blaualge Oscillatoria rubescens DC in den österreichischen Alpenseen. - Carinthia II, 163:317-330.

Helbig,J., H. Hirschwehr, E. Horsthemke & J. Schneider, 1985. Preliminary results of sedimentological investigations of two selected cores from Mondsee. - In: Contributions to the palaeolimnology of the Trumer Lakes (Salzburg) and the lakes Mondsee, Attersee and Traunsee (Upper Austria). (Eds. D. Danielopol, R. Schmidt & E. Schultze), 84-88. Limnologisches Institut, Österr. Akad. Wiss.

Jagsch,A., 1979. Veränderungen im Zustand des Mondsees in den Jahren 1968 - 1978. In: Reinhaltungsverband Mondsee. Festschrift.

Klee,R. & R. Schmidt, 1987. Eutrophication of Mondsee (Upper Austria) as indicated by the diatom stratigraphy of a sediment core. - Diatom Res. 2:55-76.

Loehr,R.C., C.S. Martin & W. Rast (Eds.), 1980. Phosphorus management strategies for lakes. - 490pp., Ann Arbor Sci. Publ. Inc., Michigan.

Moog,O., 1987. Österreichisches Eutrophieprogramm II (1983-1986). Projekt Salkammergut. Teilendbericht, Univ. Bodenkultur, Wien.

Müller-Jantsch,A., 1977. Untersuchungen an der Mondseeache als Verbindung eines eutrophen Sees mit einem oligotrophen See und Sedimentationsmessungen. - Attersee:Vorläufige Ergebnisse des OECD-Seeneutrophierungs- und des MaB-Programms 1977:52-62, OECD-Labor Weyregg.

Oberrosler,I.E., 1979. Der einfluß des Phytoplanktons auf die Fütterung von Jungfischen (Karpfen). - 137 S., Diss. Univ. Salzburg.

OECD, 1980a. OECD Eutrophication programme. Regional Project. Alpine Lakes. (Comp. H. Fricker), 233pp., Swiss Fed.Bd.Environ.Protec.,Bern.

OECD, 1980b. Cooperative programme for monitoring of inland waters. Regional project. Shallow lakes and reservoirs. (Comp. J. Clasen), 289pp., Water Res. Ctr., Medmenham.

OECD, 1980c. Eutrophication of waters. Monitoring, assessment and control. 154pp., OECD Publ. Office, Paris.

Sampl,H., 1980. Untersuchungen zum Natrium- und Chloridgehalt einiger Kärtner Seen. - Carinthia II, 90: 533 - 547.

Sampl,H., 1988. Kärntner Umweltschutzbericht 1988. 253 Seiten, Amt der Kärntner LR, Klagenfurt.

Schmidt,R., J. Müller & J. Froh, 1985. Laminated sediments as a record of increasing eutrophication of Mondsee. - In: Contributions to the palaeolimnology of the Trumer Lakes (Salzburg) and the lakes Mondsee, Attersee and Traunsee (Upper Austria). (Eds. D. Danielopol, R. Schmidt & E. Schultze), 122-131, Inst. Limnologie, Österr. Akad. Wiss.

Schultze,E., 1985. Carotenoids from selected cores of the Trumer Lakes and the Mondsee (trophic development and human impact). - In: op. cit.

Schwarz,K., 1979. Das Phytoplankton des Mondsees 1978. Arb. Lab.Weyregg 3:83-92.

Schwarz,K., 1981. Das Phytoplankton des Mondsees 1980. Arb. Lab.Weyregg 5:110-118.

Steinberg, Ch. & A. Schrimpf, 1982. Auswirkung einer künstlichen Volldurchmischung auf das Geschehen im Fischkaltersee (Osterseengebiet). - In: Beiträge zur Limnologie bayerischer Seen (Hg. Bayerisches Landesamt für Wasserwirtschaft), 7 - 54, Bayer. LA Wasserw., München.

Stumm,W., Ed., 1987. Conference on Lake Restoration. - Schweiz.Z.Hydrol.49/2:129-274.

Uhlmann,D., Ed., 1976. Eutrophierung und Gewässerschutz. - Limnologica (Berlin) 10/2:231-635.

Vollenweider,R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. - Tech. Report, 159pp., OECD, Paris.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

POST-STOCKING SURVIVAL OF ACCLIMATED AND NON-ACCLIMATED TWO-SUMMER-OLD HATCHERY BROWN TROUT (SALMO TRUTTA L.) IN A SUBALPINE RESERVOIR IN SOUTHERN CENTRAL NORWAY

T. HESTHAGEN, J. SKURDAL*

Norwegian Institute for Nature Research, Fish Research Division Tungasletta 2, 7004 Trondheim, Norway
*Environmental Division, Statsetatenes Hus
2600 Lillehammer, Norway

ABSTRACT

Two-summer-old (1+) hatchery brown trout were acclimated in a net cage for about two weeks before their release in a subalpine reservoir in southern central Norway in two successive years (1986 and 1987). Fish in the control group were released directly into the reservoir. Gill net fishing in the fall of 1987 resulted in a significantly higher recapture of acclimated than non-acclimated fish. There was no significant difference in the fraction of acclimated fish obtained from the 1986 and 1987 stocking. Age frequency distribution of non-acclimated fish marked and released in the reservoir over the past 10 years revealed few fish older than three years (3+).

INTRODUCTION

Fish used for stocking are usually captured in the hatchery, loaded into trucks, transported, and released directly at the stocking site (Barton et al. 1980). The fact that fish are chronically and acutely stressed by physical disturbance such as handling, confinement, transport and anaesthesia is well documented (Wedemeyer 1972, Soivio et et. 1977, Mazeaud et al. 1977, Strange et al. 1978, Barton et al. 1980, Pickering et al. 1982). However, less information is available about the time needed for recovery and return to normality (Pickering et al. 1982), as well as the effects of such treatments on subsequent survival after release in a natural environment (Hansen 1988, Hansen and Jonsson 1988). There are few studies on brown trout acclimation prior to release in lakes. Hesthagen and Johnsen (1989) found a higher lake survival for hatchery brown trout that were pre-stocked in a pond for about two months, than for fish that were released directly. Some positive effects on more short term acclimation have been documented for salmonids stocked in flowing water (Miller 1954, Cresswell and Williams 1983). Acclimation is defined as adaptation or adjustment to a new environment (Schmidt-Nielsen 1975).

In the present study, two-summer old (age 1+) brown trout were acclimated for about two weeks in a net cage before being released into a subalpine reservoir (1986 and 1987), while control groups were stocked directly into the reservoir. The post-stocking survival was investigated by means of gill nets. We also verified the survival of non-acclimated fish through the age-frequency distribution of hatchery fish which have been marked and released in recent years

STUDY AREA

The study was carried out in the lakes Kaldfjorden and Øyvatnet, which together with Lake Sandvatna make up the Vinstervatna Reservoir in southern central Norway. The reservoir is located in a subalpine region at an altitude of 1019m. These lakes were regulated in 1955, and the annual water level fluctuation for Kaldfjorden and Øyvatnet is 5.5 and 5.4m, respectively. The total surface area of the two lakes is 1020ha at maximum water level. The reservoir is shallow with a predominant water depth < 10 m. The water is oligotrophic with a mean specific conductivity of 14 uS cm^{-1}, and mean pH is 6.5. The regulated catchment area of 702 km^2 consists of granite and gneisses, and the vegetation cover in the vicinity of the reservoir is predominantly moorland, brush and birch.

Brown trout was initially the only fish species native to these lakes. Kaldfjorden and Øyvatnet had inflowing and outflowing streams which provided brown trout with suitable spawning and nursery areas. After the regulation in 1955, natural reproduction was seriously affected in both of these lakes. However, in Lake Sandvatna, brown trout may still spawn in the inlet as well as in several tributaries, and the brown trout stock in Kaldfjorden and Øyvatnet lakes is thought to be maintained by immigration from this area of the reservoir.

Minnow Phoxinus phoxinus, and whitefish Coregonus lavaretus were accidently introduced into the reservoir in the early 1970's (Hesthagen and Gunnerød 1980). At present, both species are considered to have established dense populations.

METHODS

About 4000 two-summer old hatchery trout were released in Kaldfjorden and Øyvatnet each year in 1986 and 1987. The mean lengths of the stocked fish in these two years were 137 mm (S.D.=14, N=544) and 123 mm (S.D.=12, N=100), respectively. In 1986, fish of the Tunhovd strain were used, while the Bjornesfjord strain was used in 1987. All fish were raised at the Reinsvoll Hatchery in circular plastic tanks, and fed on dry pellet.

The Bjornesfjord strain has been raised at the hatchery since 1966, while fish from the Tunhovd strain originated from wild parents.

One to two weeks before release, the fish were anesthetized with chlorobutanol and marked by removing the adipose fin and about 2 mm of the left and right maxilla, respectively. There were about 2000 specimens in each group. Each group of fish was transported from the hatchery to the Vinstervatna Reservoir (5h, 200 km) in separate tanks each containing 1200 l of oxygen supplied water. Fish in the control group were counted and hauled into small tanks in a boat, and subsequently released close to the shore in the Kaldfjorden and Øyvatnet lakes. Fish in the experimental group were transferred to a net cage (2x2m, depth 3m, mesh size 7mm) anchored in the reservoir. I 1987 fish were fed on pellet during their stay in the cage. The fish were counted at the start and at the end of the recovery period. The cage was covered by 0.5 cm mesh size gill nets to exclude avian predators. The fish were acclimated for a period of 9 and 14 days before release in 1986 and 1987, respectively, and the fish were stocked in the same manner as the contros specimens. The stocking date for acclimated and control fish each year is shown in Table 1. Mortality did not occur during transportation in 1986, while 7 specimens died in 1987. The mortality in the net cage was also higher in 1987 (37 specimens, 2.3%) than in 1986 (10 specimens, 0.5%). The carcasses in the net cage were difficult to count, indicating that death occurred soon after fish introduction. Unequal numbers of acclimated and control fish were adjusted for in all tests.

Table 1. Number of hatchery brown trout acclimated and stocked directly (control group) in the Vinstervatna Reservoir in 1986 and 1987, and the number of dead specimens during transportation and in the net cage. Date of stocking and duration of the recovery period is shown in parenthesis.

	Number of stocked fish		Number of dead fish	
Year	Control	Acclimated	Transport	Cage
1986	2000 (1.9.)	1897 (18.8.-27.8.)	0	10
1987	1879 (4.8.)	1946 (20.7.- 4.8.)	7	37

From 1978 to 1985, 4000 two-summer old hatchery trout (Bjornesfjord strain) were also stocked annually, but none of these were acclimated. The fish were usually marked by removing the adipose fin and 2 mm of either the left or right maxilla.

Sampling was carried out in the reservoir during three periods in the fall of 1987: 5-8 August, 1-3 September and 29-30 September. We used series of standard bottom gill nets with bar meshes between 21-45 mm, which catch trout between 19-45 cm in length with approximately the same efficiency (Jensen 1977). In addition, we used gill nets of mesh sizes 10, 12.5 and 16.6 mm. Only those age-classes which had reach a critical mean length of minimum 19 cm were used in the mortality estimate (Z):

$Z = - \ln (N_{t+1} - N_t)$.

where N_{t+1} and N_t = number of fish in age group t+1 and t, respectively (Ricker 1975).

Each fish caught was examined for marks, total length (0.1 cm), weight (1 g), and a scale sample and otoliths, which were later used for age determination were removed (Jonsson 1976).

RESULTS

For brown trout stocked in 1986 and recaptured after one year, the fraction of acclimated fish was significantly higher than for fish which were released directly ($t_s = 9.0$, df = 1, $P < 0.005$, N = 43 and 20, respectively). Among fish stocked in the fall of 1987 and recaptured 1-2 months later, a significantly greater number of acclimated than control fish were also recaptured (testing equality of two percentages, $t_s = 4.30$, df = 1, $P < 0.05$, N = 51 and 31, respectively). The number of acclimated relative to control fish was not significant among fish stocked in 1986 and 1987 ($X^2 = 0.97$, df = 1, $P < 0.05$).

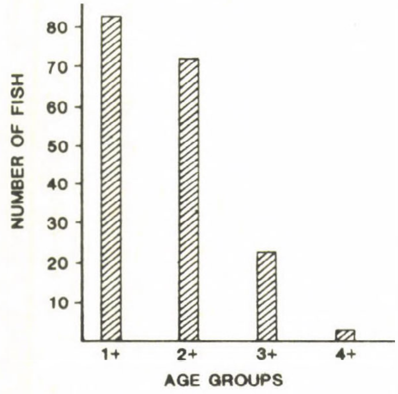


Fig. 1. Age frequency distribution of hatchery brown trout caught in the Vinstervatna Reservoir.

The age frequency distribution of hatchery fish (N=183) shows that no fish were older than 4+ (Fig. 1). The estimated annual total mortality rate (Z) was:

$$Z = 1.38 \ (SE=0.26), \ F_{1,1} = 27.63, \ P = 0.11, \ R^2 = 0.97$$

DISCUSSION

Preliminary data from the brown trout stocking program in Vinstervatna Reservoir indicates that an acclimation period of about two weeks prior to release increases subsequent lake survival compared with fish released directly into the reservoir.

The mortality of hatchery brown trout during transportation and in the net cage was relatively low, being only 0.5 and 2.3% in 1986 and 1987, repectively. However, these results suggest that the physiological status of the fish was disturbed. From a number of experiments, it is known that confinement, crowding, handling and transport of fish induce stress as measured by elevated cortisol levels which may last for weeks (Simpson 1976, Pickering et al. 1982, Pickering and Stewart 1984). Preliminary data suggests that there was disturbance in the the water - and salt balance also of the fish stocked in Vinstervatna Reservoir (M. Staurnes pers. comm.).

Hatchery fish stocked in natural waters are also subjected to malnutrition after stocking (Ersbak and Haase 1983, Bachman 1984). Hesthagen and Johnsen (1989) found significantly higher lake survival among hatchery brown trout which were pre-stocked in a pond feeding on natural food than for fish which were released directly. This source of mortality may have more negative consequences for fish released directly into a lake than for fish adjusted to the new environment in a net cage before release.

There was a high mortality among two-summer old hatchery brown trout stocked in Kaldfjorden and Øyvatnet lakes from 1978 to 1985, and few fish reached an age greater than three years.

Ability to compete and survive in natural environments is apparently reduced in hatchery fish after stocking (Miller 1954, 1958). Ayles (1975) suggested that genotype-environmental interactions for fish raised in captivity before release into the wild may be one factor contributing to the failure of many stocking programs. With the exception of those used in 1986, the stocked trout used in this study were an inbred strain. A decrease in the genetic variability seems to occur among cultured populations of salmonids (Allendolf and Phelps 1980). Ayles (1975) found that environmental differences (lake effect) accounted for most of the variation in growth and survival of rainbow trout in aquaculture lakes, but genetic and genotype-environment interactions were also significant. The lakes where the lowest survival rates were those inhabited by another fish species. Among environmental factors in the present study, competition with native brown trout and a dense population of whitefish, should be considered. There are still a heavy predominance of native brown trout among older age-classes compared with that of hatchery brown trout (Hesthagen and Skurdal 1988). Among lake trout, large native fish have been discovered to adversely affect the stocking success of hatchery fish (Purych 1977, MacLean et al. 1981). Whitefish have now established a dense population in the Vinstervata Reservoir. In Canadian lakes it is suggested that coregonids were responsible for the poor survival of introduced lake trout (Powell et al. 1986, Gunn et al. 1987).

ACKNOWLEGDEMENTS

This study was financed by the research program "Environmental effects of water regulation schemes" sponsored by The Norwegian Council for Technical and Scientific Research, and Vassdragsregulantenes Forening. Bror Jonsson gave valuable comments on a draft of this paper, Amy Lightfoot has corrected and improved the language and Finn Hellebergshaugen assisted in the field.

REFERENCES

Allendorf, F.W. and Phelps, S.R., 1980. Loss of genetic variation in a hatchery stock of cutthroat trout. Trans. Am. Fish. Soc. 109, 537-543.

Ayles, G.B. 1975. Influence of the genotype and the environmental on growth and survival of rainbow trout (Salmo gairdneri) in central Canadian lakes. Aquacult. 6,181-188.

Bachman, R.A.1984. Foraging behavior of free-ranging wild and hatchery brown trout in a stream.Trans. Am. Fish. Soc. 113,1-32

Barton, B.A., Peter, R.E. and Paulencu, C.R. 1980. Plasma cortisol levels of fingerling rainbow trout (Salmo gairdneri) at rest, and subjected to handling, transport and stocking. Can. J. Fish. Aquat. Sci. 37,805-811.

Cresswell, R. C. and Williams, R. 1983. Post-stocking movement and recapture of hatchery-reared trout released into flowing water - effect of prior acclimation to flow. J. Fish Biol. 23,266-276.

Ersbak, K. and Haase, B.L. 1983. Nutritional deprivation after stocking as a possible mechanism leading to mortality in stream-stocked brook trout. North Am. J. Fish. Mgmt.3, 142-151.

Gunn, J.M., McMurtry, M.J. and Bowlby, J.N. 1987. Survival and growth of stocked lake trout in relation to body size, stocking season, lake acidity, and biomass of competitors. Trans. Am. Fish. Soc. 116,618-627.

Hansen, L.P. 1988. Effects of Carlin tagging and fin clipping on survival of Atlantic salmon (Salmo salar L.) released as smolts. Aquacult. 70,391-394.

Hansen, L.P. and Jonsson, B. 1988. Salmon ranching experiments in the River Imsa: effects of dip-netting, transport and chlorobutanol anaesthesia on survival. Aquacult. (In press).

Hesthagen, T. and Gunnerød, T. B. 1980. Fiskeribiologiske undersøkelser i Kaldfjorden, Øyvatnet og Øvre Hersjø i Vinstra vassdraget, Oppland Fylke. DVF Reguleringsundersøkelsene. Rapp. 3-1980. 48 pp. (In Norwegian).

Hesthagen, T. and Johnsen, B.O. 1989. Lake survival of hatchery and pre-stocked pond brown trout, Salmo trutta L. Aquacult. Fish. Mgmt. 20 (In press).

Hesthagen, T. and Skurdal, J. 1988. Akklimatisering av to-somrig settefisk av aure før utsetting. Miljøvirkninger av vassdragsutbygging. Rapp. nr. B44. 10 pp. (In Norwegian).

Jensen, K.W. 1977. On the dynamics and exploitation of the population of brown trout, Salmo trutta L., in Lake Øvre Heimdalsvatn, Southern Norway. Rep. Inst. Freshw. Res.Drottningholm 56,18-69.

Jonsson, B. 1976. Comparison of scales and otoliths for age determination in brown trout, Salmo trutta. Norw. J. Zool.24,295-301.

MacLean, J.A., Evans, D.O., Martin, N.V. and DesJardine, R.L. 1981. Survival, growth, spawning distribution, and movement of introduced and native lake trout (Salvelinus namaycush). Can. J. Fish. Aquat. Sci. 38,1685-1700.

Mazeaud, M.M. Mazeaud, F. and Donaldson, E. M. 1977. Primary and secondary effects of stress in fish, some new data and a general review. Trans. Am. Fish. Soc. 106,201-212.

Miller, R. B. 1954. Comparative survival of wild and hatchery reared cutthroat trout in a stream. Trans. Am. Fish. Soc. 83,120-130.

Miller, R.B. 1958. The role of competition in the mortality of hatchery trout. J. Fish. Res. Bd Can. 15,27-45.

O'Grady, M. F. 1983. Observations on the dietary habits of wild and stocked brown trout, Salmo trutta L., in Irish lakes. J. Fish Biol. 22,593-601.

Pickering, A. D., Pottinger T.G. and Christie, P. 1982. Recovery of the brown trout, Salmo trutta., from acute handling stress: a time-course study. J. Fish Biol. 20,229-244.

Pickering A.D. and Stewart, A. 1984. Acclimation of the interrevial tissue of the brown trout, Salmo trutta., to chronic crowding stress. J. Fish Biol.2,731-740.

Powell, M.J., Bernier, M.F., Kerr, S.J., Leering, G., Miller, M., Samis, W. and Pellegrini, M. 1986. Returns of hatchery-reared lake trout from eight lakes in northeastern Ontario. Ontario Ministry of Natural Resources, Toronto, Ontario Fish. Tech. Rep. Ser. 22.

Purych, P.R. 1977. Poor returns of hatchery-reared lake trout to the sport fishery of Flack Lake, Ontario. Prog. Fish-Cult. 39,185-186.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Bd Can. 191. 382pp.

Schmidt-Nielsen, K. 1975. Animal physiology. Adaptation and environment. Cambridge University Press. 699pp.

Simpson, T.H. 1976. Endocrine aspects of salmonid culture. Proc. R. Soc. Edinb. 75B,241-252.

Soivio, A., Nyholm, K. and Huhti, M. 1977. Effects of anaesthesia with MS 222, neutralized MS 222 and benzocaine on the blood constituents of rainbow trout, Salmogairdneri. J. Fish Biol. 10,91-101.

Strange, R.J. Schreck, C.B. and Ewing, R.D. 1978. Cortisol concentrations in confined juvenile chinook salmon (Oncorhynchus tshawytscha). Trans. Am. Fish. Soc. 107,812-819.

Wedemeyer, G. 1972. Some physiological consequence of handling stress in the juvenile coho salmon (Oncorhynchus kisutch) and steelhead trout (Salmo gairdneri). J. Fish. Res. Bd Can. 29,1780-1783.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

PHYTOPLANKTON COMMUNITIES IN 151 SMALL LAKES OF DIFFERENT TROPHIC STATUS AND WITH DIFFERENT FORMS OF LAND-USE IN THEIR CATCHMENT AREAS

V. ILMAVIRTA, P. HUTTUNEN *

Maj and Tor Nessling Foundation, Hesperiankatu 3 A 4,
SF-00260 Helsinki, Finland
* Department of Biology, University of Joensuu
P.O. Box 111, SF-80101 Joensuu, Finland

ABSTRACT

Phytoplankton assemblages, water chemistry and land-use parameters are compared as a part of a long-term study on 151 lakes in eastern Finland. The lakes are grouped by PCA and stepwise regression analysis in order to describe significant variations in parameters.

INTRODUCTION

The water quality of the lakes of the major watercourses in Finland is largely determined by water discharged from numerous chains of small lakes. Most of these are surrounded by coniferous forests or peatlands, which are the main sources of the brown coloration in the water. Many lakes influenced by intensive agriculture or forestry and settlement are meso-eutrophic, or even hypereutrophic, and some lakes are also naturally acidic. Most of the above lake types are represented in a long-term research programme started in 1978 regarding 151 lakes in eastern Finland. The work has been supported by grants from the Maj and Tor Nessling Foundation and the Academy of Finland. The main goals are to investigate in a comprehensive manner the relations between water chemistry, land-use and biological assemblages in lakes and evaluate the community ecology of the water organisms as a tool for analysing the character of lakes. Attempts are also being made to find principles for the classification of lakes on the basis of simple chemical and biological parameters for the purpose of water quality monitoring and environmental planning. The research strategy is presented in more detail in MERILÄINEN et al. (1981). The relations between certain taxonomical groups and lake water properties have been reported earlier, i.e. those involving surficial diatom assemblages (MERILÄINEN & HUTTUNEN 1984, HUTTUNEN & MERILÄINEN 1986 a,b,c), surficial Cladoceran assemblages (COTTEN 1985, HUTTUNEN et al. 1988), surficial Chrysophyta scales (CHRISTIE et al. 1988) and phytoplankton (K. ILMAVIRTA et al. 1984, ILMAVIRTA 1984, 1988). The paper is an extended abstract covering the large set of data from all 151 lakes and the results will be discussed in greater detail later in a number of more specialized papers.

THE AREA STUDIED

The present paper is based on data for 151 lakes located partly on the granite complex and partly on Karelidic schists or apatite bedrock in eastern Finland. Small or medium sized headwater lakes were selected for the study in order to avoid watercourse lakes and lakes with a heavy point-load. More than 60 physico-chemical parameters were measured regarding the water, bottom sediments and catchments areas. The sediment results are not included in this paper.

Most lakes were small, < 150 ha (n = 121), while 22 were of size 150 - 500 ha and only 8 lakes greater than 500 ha. Maximum depth averaged 13.3 ±0.7 m, 62 % of the lakes being shallower than 15 m throughout, 32 % 15 - 30 m at their deepest point and only 3 over 40 m. All lakes were stratified in summer (dimictic), the depth of the epilimnion varying from 1 to 9 m.

The catchment area (excluding the lake area) ranged from 0.2 to 179 km^2. Most were small: 108 smaller than 8.2 km^2 and only 4 greater than 100 km^2. In 125 cases the catchment area was covered by coniferous forests, fields being dominant land-use form around 4 lakes, peatlands around 15 lakes. The minimum, average and maximum proportions of fields were 0 %, 11.9 % and

71 %, those of peatlands 0%, 18.1 % and 80 %, and those of forests 19 %,70 % and 100 %. The percentage of shoreline bounded by fields (fshore) was measured. Few lakes were entirely surrounded by fields (134 lakes had less than 50 % such shoreline). The average fshore index was 20.1 ±26.5 %, the minimum 0 % (42 lakes) and the maximum 95 % (2 lakes).

METHODS

Water samples were collected from the middle of each lake twice: during the summer stagnation period (first sample 13.7.1980, last sample 12.8.1980) and during the autumnal overturn 1978 (50 lakes) and 1979 (101 lakes). Each water sample per lake consisted of four to ten independent subsamples (volume 6.4 litres) taken through the epilimnion in summer and from 1 - 2 m in autumn. This large, composite sample was carefully mixed before performing the analyses. Composite samples of this kind have been shown to represent well the mean water quality of the open water area of a lake (ILMAVIRTA 1980). Water for the chemical analyses was transferred to the laboratory in polyethene bottles stored in dark coldboxes. The phytoplankton samples were fixed with acetic Lugol solution in the field . The water samples were analysed by standard methods used by the National Board of Waters in Finland (ERKAMAA et al. 1977).

Phytoplankton was counted in sedimentation chambers with the inverted microscope (UTERMÖHL 1958). More than 500 items (cells, colonies, 100 μm filaments) were counted in random visual fields. This method provided a good estimate of cell numbers, biomass and species composition by groups, but underestimates markedly the number of species in the sample. Biomass was calculated using standard individual cell volumes (NAULAPÄÄ 1977 with numerous corrections and additions). The taxonomy used is according to the "Guide to Phytoplankton" by TIKKANEN (1986). The microscope analyses were performed by Mrs. Kaija Ilmavirta, Phil. Cand., to whom thanks are due for this work.

Principal component analysis (PCA) with varimax rotation was used to group the lakes by reference to the data on autumnal water chemistry. The relationships between the biological and environmental data bases were analysed by multiple regression analysis with stepwise selection of variables.

RESULTS AND DISCUSSION

Due to the large set of data studied the variation both in water chemistry and land-use parameters and in phytoplankton assemblages was high and it was difficult to find clear linear correlations between variables. Therefore, the water chemistry data were further processed by PCA to obtain more homogeneous groups of lakes. The chemical data for the autumn were used for this purpose on the assumption that they describe better the more or less stable average chemical balance in the lakes than does the rapidly changing situation in the lakes during the most active production period in summer. The PCA method was chosen because use of a moving average or smoothing of the data in this method will dissipate a significant part of the individual variation. PCA axes 1, the "Acidity axis" and 2, the is "nutrient and colour axis" (Fig.1) together explained 75.6 % of the total variance. The main characteristics of the groups and the numbers of lakes are as follows:

A. Acidic oligotrophic clear water lakes (9)
B. Sensitive oligotrophic clear water lakes (6)
C. Sensitive oligotrophic clear water lakes, turbid (21)
D. Sensitive oligotrophic mesohumic lakes (31)
E. Acidic mesotrophic polyhumic lakes (16)
F. Neutral oligotrophic clear water lakes (15)
G. Buffered oligotrophic clear water lakes (5)
H. Buffered oligo-mesotrophic clear water lakes (17)
I. Neutral mesotrophic mesohumic lakes (17)
J. Neutral eutrophic polyhumic lakes (10)
K. Buffered eutrophic mesohumic lakes (3)

Many water properties of the lake groups still overlap, but some groups already differ significantly from others, as seen in the notched box-and-whisker plots (Fig. 2). The stepwise multiple

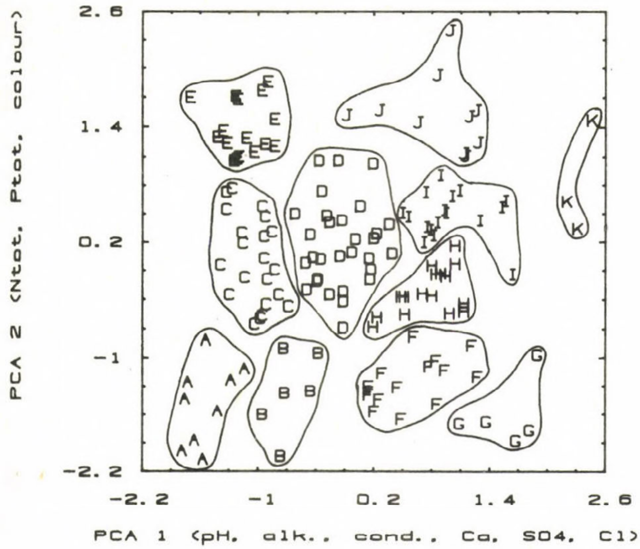


Fig. 1. PCA groups (A-K) formed from the 151 lakes studied in eastern Finland. Grouping is based on all the chemical parameters measured for the autumn samples. The most heavily weighted parameters are given.

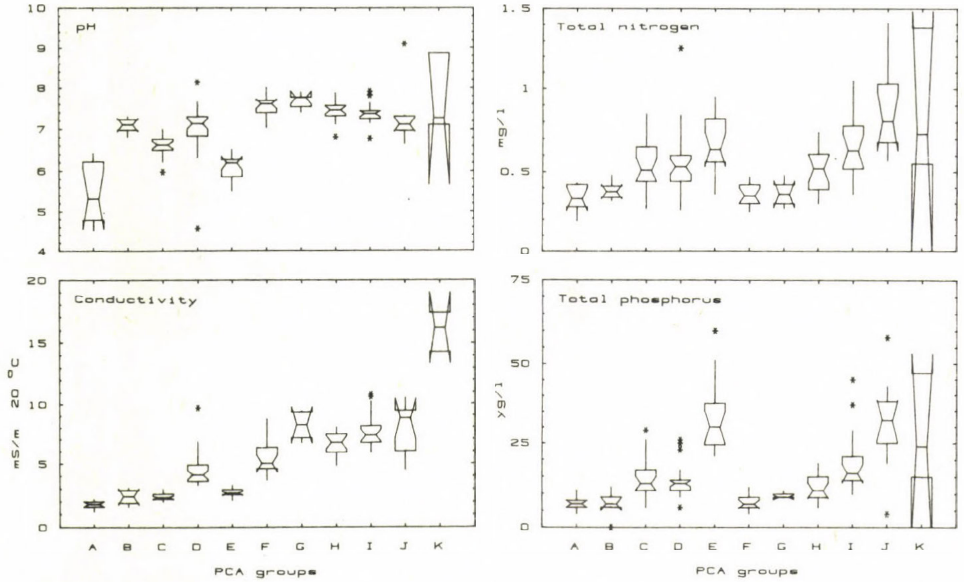


Fig. 2. Notched Box-and-Whisker plot for pH, total nitrogen, conductivity and total phosphorus in summer in the PCA lake groups.

regression analyses using average catchment characteristics explained well the variation in many of the chemical parameters, as given below (DEPENDENT VARIABLE: independent variables; variation explained):

NTOT: forest (-), fshore	96 %
PTOT: forest (-)	88 %
SECCHI: forest	82 %
pH: peatland (-)	80 %
NO_3: fshore, catchment area	75 %
CONDUCTIVITY: field	65 %
(ALKALINITY: field	44 %)
(Ca: none	0 %)

The lake area and in most cases also the catchment area has no significant role as an explanatory variable. The proportion of peatlands was only significantly in explaining the pH of the water.

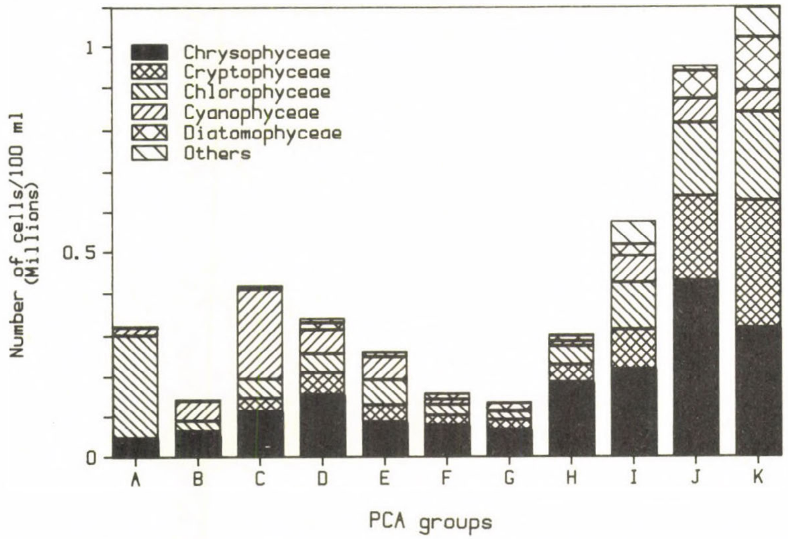


Fig. 3. Numbers of cells of given taxa in lake groups A-K. Euglenophyceae, Dinophyceae and Zygnematales are not presented because of their low occurrence.

The highest mean biomasses and numbers of phytoplankton cells (Fig. 3) were recorded in lake groups K, J and I, while the lowest biomass was found in the nutrient-poor acid group B and the lowest number of cells in group G, with high pH. The correlation between biomass and number of cells was high (r^2 = 0.94) in spite of the differences in species composition. Since biomass figures must be considered to be partly estimates due to possible errors in individual volume measurements, the numbers of cells are used in the following discussion.

Chrysophytes were dominant in all the lake groups except groups A and C. The next most common were Chlorophyceae or Cryptophyceae in groups C-K, but in group A Chrysophyceae and Chroococcales blue-green algae in group B. Dinophyceae was sparse in all the lake groups, and cell numbers of Euglenophyceae and Dinophyceae were low, and were lacking in most groups, indicating clearly the oligotrophic character of the present lakes and the absence of pollution by sewage load.

The variations in total biomass, cell numbers, chlorophyll-a concentration and the proportions of many of phytoplankton taxa were explained fairly well by the water parameters using multiple regression analysis with stepwise selection of variables. The results of analyses based on the summer measurements are given below (DEPENDENT VARIABLE: independent variables, variation explained):

TOTAL BIOMASS: Ntot, pH, alkalinity (-)	98%
TOTAL CELL NUMBER: Ntot, colour (-), Ca	97%
CHLOROPHYLL a: Ntot, pH, conductivity (-)	99%

Cell numbers in taxonomical groups:

CRYPTOPHYCEAE: Ntot, pH, Secchi	97%
CHRYSOPHYCEAE: Ntot, pH, conductivity (-), alkalinity	97%
DIATOMOPHYCEAE: Conductivity, alkalinity (-)	94%
DINOPHYCEAE: conductivity, alkalinity (-), pH	84%
NOSTOCALES: Ntot, pH	79%
CHROOCOCCALES: pH (-), Ca	74%
(CHLOROPHYCEAE: none	0%)

Total nitrogen seems to be the most important variable determining cell number distribution in the lakes studied. There is an obvious autocorrelation between biomass and total nitrogen, since nitrogen accumulates in the biomass during the growing season and its recycling time is longer than that of phosphorus. This is acceptable, however, because we are attempting to find a chemical parameter which is easily measured to describe the changes in the structure and functioning of the phytoplankton communities in the lakes.

pH is an important variable for many algal groups containing small-celled species, but variables describing the mean nutrient balance (conductivity) and resources for primary production (alkalinity) are more important for algae bearing hard cell covers. It is also significant that the occurrence of Chlorophyceae species is weakly explained even by the average PCA variables. A more precise approach is needed for this purpose. It is similarly important that Si does not play any role in the occurrence of diatoms in spite of significant differences between the summer and autumn concentrations. Euglenophyceae occurred so sparsely that no calculations could be performed, and the Zygnematales were likewise omitted because of their low cell numbers.

REFERENCES

Christie, C., J. Smol, P. Huttunen & J. Meriläinen 1988. Chrysophyte scales recorded from eastern Finland. Hydrobiologia 161: 237-243.

Cotten, C., 1985. Cladocera assemblages related to lake conditions in eastern Finland. Indiana Unic. (U.S.A.), Dept. Biol., Doctoral thesis. 97 pp.

Erkamaa, K., J. Mäkinen & O. Sandman, 1977: Methods of water analysis by authorized and water authority laboratories (in Finnish). Nat. Bd Waters, Rep. 121: 1-54.

Huttunen, P. & J. Meriläinen, 1986a. Applications of multivariate techniques to infer limnological conditions from diatom assemblages. Devel. Hydrobiol. 29: 201-211.

Huttunen, P. & J. Meriläinen, 1986b. A comparison of different pH indices derived from diatom assemblages. Univ. Joensuu, Publications of Karelian Institute 79: 41-46.

Huttunen, P. & J. Meriläinen, 1986c. Diatom response to pH and humic matter of the water. Univ. Joensuu, Publication of Karelian Institute 79: 47-54.

Huttunen, P., J. Meriläinen, C. Cotten & J. Rönkkö 1988. Attempts to reconstruct lake water pH and colour from sedimentary diatoms and Cladocera. Verh. Internat. Verein. Limnol. 23: 870-873.

Ilmavirta, K., P. Huttunen & J. Meriläinen, 1984. Phytoplankton in 151 eastern Finnish lakes: species composition and its relations to the water chemistry. Verh. Internat. Verein. Limnol. 22: 822-828.

Ilmavirta, V., 1980. Phytoplankton in 35 Finnish brown-water lakes of different trophic status. Devel. Hydrobiol. 3: 112-120.

Ilmavirta, V., 1984. The ecology of phytoplankton in brown-water lakes. Verh. Internat. Verein. Limnol. 22: 817-821.

Ilmavirta, V., 1988. Phytoplankton and their ecology in Finnish brown-water lakes. Hydrobiologia 161: 255-270.

Meriläinen, J., P. Huttunen & K. Pirttiala 1981. Research strategy for the ecology of diatoms: the diatom study of 151 lakes in eastern Finland. In, R. Ross (ed.), Proceedings of the sixth symposium on recent and fossil diatoms, Otto Koeltz, Koenigstein: 413-423.

Meriläinen, J., P. Huttunen, 1984: Ecological interpretation of diatom assemblages by means of two-way indicator species analysis (TWINSPAN). In, D.G. Mann (ed.), Proceedings of the seventh diatom symposium: 385-391.

Tikkanen, T., 1986. Kasviplanktonopas. (Guide to phyto-plankton). In Finnish. Suomen Luonnonsuojelu Tuki OY, Forssa. 278 pp.

Utermöhl, H., 1958. Zur Vervollkomnung der quantitativen Phytoplanktonmethodik. Mitt. Internat. Verein. Limnol. 9: 1-38.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

CHARACTERISTICS OF THE LAKE LADOGA ECOSYSTEM

I.M. RASPOPOV

Institute of Limnology of the Academy of Sciences
of the USSR
ul. Sevastyanova 9, Leningrad, 196199, USSR

Lake Ladoga is the largest body of freshwater in Europe with a surface area of 18,135 km^2 including islands of an area of 460 km^2. The volume of the lake is 908 km^3, its average and maximum depths being 51 and about 230 m, respectively. Mean depth of the shallowest southern part is 13 m.

Lake Ladoga is situated on the borderline of the crystalline Baltic shield and the Great Russian Plain. The geological history of the drainage basin of 250,600 km^2 surface area has been summarised by Arkhangelskií (1947, 1948). Differences in the geological structure of the watershed are reflected in the structure of both the shores and the depressions of the lake.

Seven types of bottom sediments were distinguished by N.I. Semenovich (1966): 1. blocks, 2. boulders, 3. pebbles and gravel, 4. sand of various grain size, 5. coarse-grained aleurite silt, 6. fine-grained aleurite silt, and 7. clayey silt. Clayey silt accumulates in the deepest areas of the lake. The other types of bottom sediments are characteristic of the littoral and sublittoral zones. The organic matter content of the sediments is low remaining below 3.4% even in the fine-grained aleurite silts.

The principal components of the water balance are inflow and outflow, accounting for 86 and 92% of the total inputs and outputs, respectively. Since 1981 the annual inflow has varied between 77.8 and 89.0 km^3. The average annual fluctuation of the water level is 0.7 m.

The lake is thermally stratified during the summer. The thermocline appears first along the shore and is gradually shifted towards the deepwater region. The epilimnion and the hypolimnion are distinctly different in their physicochemical characteristics (Tikhomirov 1982).

Wind induced wave action is characteristic in Lake Ladoga. The maximal wave height and length observed were 5.8 and 60 m, respectively (Malinina 1966).

A deterioration of the favourable oxygen conditions has been detected during the last 10 years induced by the anthropogenic eutrophication. The concentration of dissolved oxygen fluctuates between 9–15 mg l^{-1} corresponding to 80–120% oxygen saturation (Petrova 1982).

The concentration of total phosphorus is a widely accepted chemical indicator of the trophic level of water bodies. In the 1960s Lake Ladoga was an oligotrophic lake where the development of phytoplankton was limited by phosphorus. During 1962–1979 concentration of phosphorus has shown a sharp increase as a consequence of the elevated municipal waste water loading and rapid industrial development. Maximum values of total phosphorus (TP; 28 $\mu g\ l^{-1}$) were registered in the spring and summer of 1980 and 1981. By 1984–1985 the average TP concentration has decreased to 24 $\mu g\ l^{-1}$, due to the introduction of sewage treatment in the Volkhov aluminium plant. Inorganic phosphorus accounts for 40–60% of TP during the spring. During the summer its share is decreased to 8–18% by the intense algal assimilation. In the hypolimnion 22–40% of TP remains inorganic.

The second most important limiting nutrient is nitrogen. Lake Ladoga has always been supplied with an excess of inorganic nitrogen compounds, primarily nitrate. The concentrations of both inorganic and organic forms of nitrogen are relatively constant during the whole year. The average annual total nitrogen (TN) concentration is 0.6 mg l^{-1}.

There is a significant load of silicon and other biogenic elements through the tributaries. The present tributary silicon load is similar to that during the 1960s. However, the concentration of silicon amounting to 0.9 mg Si l^{-1} in the 1960s has dropped to half by the present time as a consequence of the intense silicon assimilation of the diatoms (Petrova 1982).

The allochthonous and autochthonous sources of organic matter have an approximately equal share in the total input of $2x10^6$ tons organic carbon per year. The average concentration of organic carbon is 10 mg C l^{-1} in the ice-free period. During the eutrophication and the period of restoration considerable spatial and seasonal changes were observed in the organic carbon concentrations (Petrova and Raspletina 1987).

Both the flora and fauna are diverse in Lake Ladoga. About 600 plant and more than 800 animal species are known from the lake (Table 1), reflecting the remarkable diversity of biotopes in the lake. The structure and dynamics of the populations are, however, more informative than mere species lists.

Table 1. Number of plant and animal species in Lake Ladoga

Community	Number of species, subspecies and forms
Phytoplankton	427
Phytoperiphyton	344
Fungi	35
Macrophytes	62
Zooplankton	378
Benthos	408
Fishes	46

Diatoms, blue-green and green algae may dominate the phytoplankton. A comparison of the composition of phytoplankton before and during the anthropogenic eutrophication (periods 1956-1962 and 1976-1979, respectively) indicates that the species composition remained essentially unchanged (Table 2, Petrova 1982, Petrova and Raspletina 1987).

Table 2. Composition of phytoplankton in Lake Ladoga

Algal groups	% of total number of algal species 1956-1962	1976-1979
Cyanophyta	20.0	19.2
Chrysophyta	3.7	3.2
Bacillariophyta	40.5	38.4
Xanthophyta	0.8	1.8
Pyrrophyta	1.8	4.2
Euglenophyta	0.8	0.4
Chlorophyta	33.2	32.8

In the early 1960s only 5 species, two blue-greens and four diatoms were occasionally dominant with cell numbers exceeding 10^6 cells l^{-1}. Anthropogenic eutrophication resulted in a sharp increase in the biomass of most algal species. Maximum number of 22 species (7 diatoms, 11 blue-greens, 3 green algae and 1 yellow-green alga) has recently exceeded 10^6 cells l^{-1} (Petrova 1982). In addition to species dominant in oligotrophic lakes, new species became dominant which require higher phosphorus concentration for their growth. Previously diatoms dominated also in the summer after the spring bloom. At the present time blooms of blue-green, yellow-green algae and diatoms are formed in the summer. Production and biomass of the principal algal groups are given seasonally in Table 3. Production of the littoral diatoms was the highest.

Since 1976-1978 the annual production has been nearly constant. During the vegetation period the production varied between 45.0-54.5 g C m^{-2} in the different years. The average annual production ranged between 800-970 x 10^3 tons of carbon.

The diverse periphyton (Table 1) is dominated by diatoms (47%) and green algae (37%). Species composition and biomass of the periphyton is primarily determined by the water dynamics. The number and biomass of Ulotrix zonata intensely

Table 3. Production and biomass of basic groups of phytoplankton in Lake Ladoga in the period 1976-1983

Algal group	Biomass μg chl-a^{-1}	Production μg C $l^{-1}h^{-1}$
	SPRING	
Aulacosira (littoral zone)	2.7 / 0.4 - 7.2	324 / 19 - 1183
Aulacosira (profundal zone)	0.2 / 0.02- 2.6	13 / 3 - 36
Aulacosira-Diatoma (littoral zone)	2.1 / 1.0 - 4.9	391 / 109 - 1050
	SUMMER	
Asterionella	1.4 / 0.5 - 11.7	45 / 13 - 210
Asterionella-Diatoma	3.5 / 1.7 - 8.2	373 / 68 - 492
Asterionella-Tribonema	2.4 / 2.3 - 2.5	123 / 74 - 173
Tribonema	1.6 / 0.6 - 3.3	180 / 49 - 452
Fragillaria	1.4 / 0.9 - 1.5	127 / 78 - 177
Oscillatoria	1.6 / 0.2 - 2.4	270 / 120 - 387
Aphanizomenon	2.3 / 0.2 - 21.3	117 / 14 - 1390
Microcystis	3.0 / 0.2 - 6.3	237 / 7 - 4220
	AUTUMN	
Woronichinia	0.6 / 0.2 - 1.0	69 / 42 - 96
Aulacosira-Tribonema	1.1 / 0.3 - 1.6	277 / 2 - 97

growing on the rocky and stony substrata of the northern part of the lake reaches $1-3x10^6$ cells cm^{-1} and 200-400 mg cm^{-2}, respectively. Total production and respiration amounted to 12.7 and 4.6-6.9 g C $m^{-2}day^{-1}$ during July-August. Ulotrix inhabits a narrow shoreline zone less than 50 cm wide. In the deeper zones the diatoms Achnanthes, Gomphonema, Fragillaria, Tabellaria become dominant with maximum biomass and chlorophyll-a content of 1 mg cm^{-2} and 1 μg cm^{-2}, respectively.

In the northern bays of the lake large fluctuations can be observed in the number ($5,300-2x10^6$ cells cm^{-2}), biomass (0.002-135.0 mg cm^{-2}) and chl-a content (0.004-70.5 μg cm^{-2}) of the diverse periphyton community.

In the southern area of the lake the average biomass and chl-a content of the periphyton reach 2.0 mg cm^{-2} and 1.8 μg cm^{-2}, respectively.

These data suggest that primary production by periphyton is not significant in Lake Ladoga.

Mycological studies were initiated in Lake Ladoga in the 1980s: 35 species of aquatic fungi have been registered, the dominant groups being Oomycetes and Dentoromycetes. The bulk of fungi is saprophytic in the lake, only three parasites were discovered. Aquatic fungi are predominantly epilimnetic with uneven spatial distribution. The number of fungi increases offshore. In August 1985 the number of fungi fluctuated between 500-5,000 cells l^{-1} (average 2,150 cells l^{-1}) in the surface layer (Petrova and Raspletina 1987).

On the basis of the characteristics of the aquatic vascular vegetation three regions can be distinguished in Lake Ladoga: the skerry region, the region of open western and eastern shores and the region of the southern bays. In the upper littoral zone Phragmites australis, Scirpus lacustris and Potamogeton perfoliatus are the dominant macrophyte species. Macrophytes cover an area of 105 km^2, with an annual production of 24,200 tons C or 1.32 g C m^{-2}. The annual production is different in the three geobotanical regions amounting to 1.9, 5.2 and 0.17 g m^{-2} in the skerry region, along the western and eastern shores and in the southern bays, respectively. Of total macrophyte production, 77% was in the southern area of the lake. The share of emergent and submersed macrophytes was 90 and 8.6% within the total production, respectively. About 30% of the organic matter assimilated by the macrophytes is not decomposed during the year. This portion is either deposited on the bottom in the form of coarse organic remnant or harvested (Raspopov 1985).

There are significant spatial differences in the number of both total and saprophytic bacteria (0.27×10^6 and 82 cells ml^{-1} in the middle of the lake and 1.77×10^6 and 455 cells ml^{-1} in the western area, respectively). The bacterial production amounted to 25.0 g C m^{-2} in the summer 1986. The amount of organic matter decomposed by bacteria was estimated at 580,000 and 370,000 tons C during the summer and autumn of 1986, respectively. Respective total production was 1.9×10^6 and 1.2×10^6 tons C.

Several zooplankton species of Lake Ladoga are widely spread over also in other temperate lakes of the Northern Hemisphere (Table 4).

Table 4. Composition of zooplankton in Lake Ladoga

Groups	Species and subspecies	
	Number	% of total
Protozoa	90	24
Rotatoria	200	53
Cladocera	61	16
Copepoda	27	7

A high diversity and number of Rotatoria is remarkable. Since the large Asplanchna periodonta is one of the dominant species, their share in the biomass of zooplankton may be even 86%. The biomass of zooplankton ranged between 6.1-19.4 g m^{-2} in August 1983 in the different areas of the lake.

No changes could be observed in the species composition of zooplankton during the eutrophication of Lake Ladoga. The

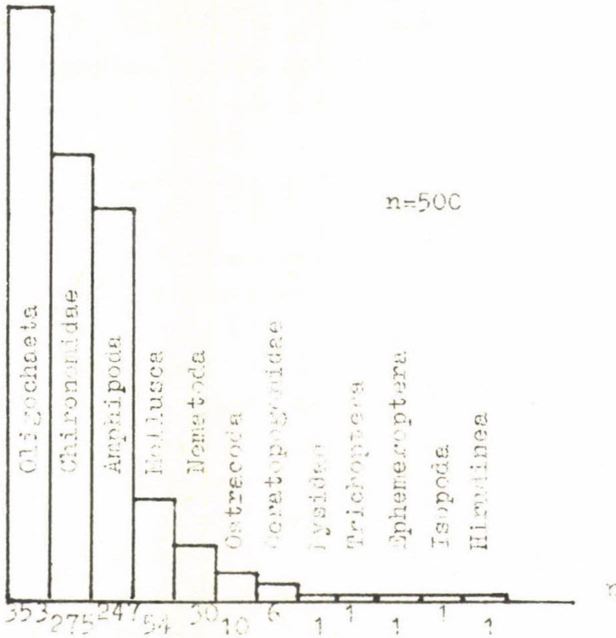


Fig. 1. Frequency of the benthos general groups

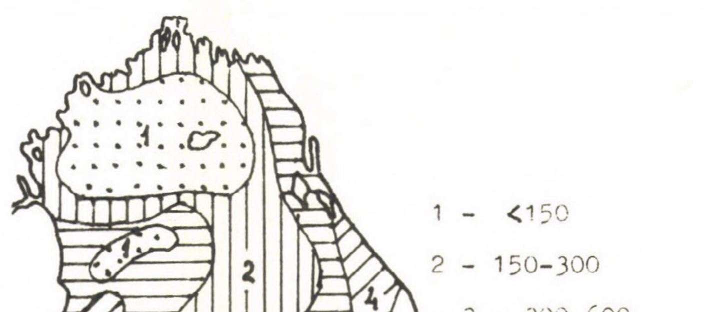


Fig. 2. Distribution of the meiobenthos biomass

structure of the zooplankton community has changed and the biomass increased. The latter was, however, influenced not only by anthropogenic factors but also by climatic ones (Petrova and Raspletina 1987).

Of macrozoobenthos 408 species and forms were identified in Lake Ladoga (Gerd 1946). Most of them inhabit different littoral biotopes. Only 15% of the species can be found in the sublittoral and the profundal zones. The fauna of the deepwater areas is especially poor.

The frequency distribution of the main macrobenthos groups is shown in Figure 1. Oligochaeta are the most widely distributed group of benthic invertebrates. They were found in 92% of samples representing different depths and types of sediments.

The biomass of the macrobenthos increases from 0.1 g m^{-2} in the deepwater region to 6.0 g m^{-2} in the silted sand of the sublittoral zone. The maximum biomass (28 g m^{-2}) was observed in the Volkhov inlet.

Meiobenthos has been sampled regularly since 1983 in Lake Ladoga. This group can be found at all depths. Five different zones could be distinguished on the basis of the increasing biomass of the meiobenthos (Fig. 2). The maximum biomass was observed again in the Volkhov inlet (125,500 ind. m^{-2} or 7.6 g m^{-2}).

The share of meiobenthos within the total biomass of the zoobenthos ranged from 2 to 70%. The minimal values (2-15%) were detected in the northern and central parts of the lake. In the western and north-eastern areas of Lake Ladoga the ratio of meiobenthos ranged between 15-30%, whereas in the southern bays and along the eastern coast it increased up to 30-70% (Petrova and Raspletina 1987).

The ichthyofauna of Lake Ladoga is represented by 46 species. Lake Ladoga is originally a water body with salmon - whitefish dominance. However, at present the mass species important for fishery are *Osmerus eperlanus*, *Coregonus albula*, *C. lavaretus*, *Stizostedion lucioperca*, *Perca fluviatilis*, *Rutilus rutilus*. The number of salmon is very low and its fishery has been banned since 1961. In order to increase the number of Salmon salar and Coregonus lavaretus, about five million fries are introduced annually into the lake by the fishery plant.

During the last 30 years the annual fishery catch fluctuated between 2,100 and 6,400 tons. In addition to this, the annual catch by anglers amounted to 2,000 tons. Taking into account both fishery and angling, the total catch runs to 9 kg ha^{-1}. The resources of fishery can be estimated at 21,000 tons.

Great attention is being paid to the preservation of the water quality in Lake Ladoga. In 1984 the Council of Ministers of the USSR adopted a resolution on protecting measures of Lake Ladoga and its basin. Implementing this resolution, the large pulp and paper plant of Priozersk was closed down in 1986 because the sewage treatment was insufficient. The governmental program "Ladoga" has been elaborated and is being carried out by the cooperation of several different institutions. The Limnological Institute of the Academy of Sciences of the USSR has been assigned to coordinate the program. Intense studies of Lake Ladoga aiming at a synthesis are in progress.

References

Arkhangelskií A.D. 1947, 1948. Geological structure and geological history of the USSR. (In Russian). Publ.House Ac.Sci. USSR, Moscow, 1, 415 p., 2, 372 p.

Gerd S.V. 1966. A review of hydrobiological studies in Karelian lakes. (In Russian). Trudy Karelo-Finskogo otdeleniya VNIORH", Petrosavodsk, 26-140.

Malinina T.I. (ed.) 1966. Hydrological regime and balance of Lake Ladoga. Publ. House Ac.Sci.USSR, Leningrad, 324 p.

Petrova N.A. (ed.) 1982. Anthropogenic eutrophication of Lake Ladoga (In Russian). Nauka, Leningrad, 304 p.

Petrova N.A. and Raspletina G.F. (eds.) 1987. Natural state of the Lake Ladoga ecosystem. (In Russian). Nauka, Leningrad, 213 p.

Raspopov I.M. 1985. Higher aquatic vegetation of large lakes in the northwestern area of the USSR. (In Russian). Nauka, Leningrad, 200 p.

Semenovich N.I. 1966. Bottom sediments of Lake Ladoga. (In Russian). Publ.House Ac.Sci.USSR, Moscow and Leningrad, 124 p.

Tikhomirov A.I. 1982. Thermal structure of large lakes. (In Russian). Nauka, Leningrad, 232 p.

Titenkov N.S. 1968. Fishes and fishery in Lake Ladoga. (In Russian). In: Biological resources of Lake Ladoga (zoology). Nauka, Leningrad, 130-173 p.

MANAGEMENT OF SHALLOW WATERS

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

THE DECAY OF REED IN HUNGARIAN LAKES

M. KOVÁCS,[1] G. TURCSÁNYI,[1] Z. TUBA,[1] S.E. WOLCSÁNSZKY,[2] T. VÁSÁRHELYI,[3] Á. DELY-DRASKOVITS,[3] S. TÓTH,[1] A. KOLTAY,[1] L. KASZAB,[1] P. SZŐKE,[1] B. JANKÓ[4]

[1]Agricultural University, Gödöllő, Hungary; [2]Teachers' Training College, Eger, Hungary; [3]Department of Zoology, Natural History Museum, Budapest; [4]Department of Botany, Natural History Museum, Budapest

In the last 30-40 years in numerous eutrophic lakes of Europe the decay of reed stands, the withdrawal of their front line has been recognized. The previously unintermittent, closed stands have become sparse and have been divided into small clumps. In consequence of the phenomenon, known as "the decay of reed", not only the area of the stands decreases, but also the industrial quality of the shoots, ripe for harvesting, declines considerably. These shoots are often much shorter and thinner than required. In Europe first of all that reed is decaying, the chromosome number of which is 48.

The literature defines a number of factors as possible causes of reed decay. Some authors suggest that mechanical effects play a great role in dividing the closed stands into clumps. Reed stands provide good possibilities for landing of boats as well as fishing, and to fulfil this function, stages have been built. In order to form such places, green shoots will be cut often under the water-surface. As a result of this water gets into the wounded rhizomes and without oxygen they begin to rot.

By the embankments, built up of either stone or concrete, the hard bottom cannot support the reed stands, neither the rhizome can take root in the loose deposit, to be found in front of these. As at the embankments the turbulence of water grows stronger, the increasing surf energy crushes even more easily the otherwise already thinned stands. The rhizomes and the buds are also injured by the mechanical reed harvesters, if the harvesting does not take place on solid ice. Amongst further mechanical injurers we must mention the coot (Fulica atra), feeding on the young shoots of reed and the muskrat (Ondatra zibethica).

According to other authors the direct and indirect effects of eutrophication play a more important role in the decay of reed, than mechanical effects. In this respect first of all the sketch of Schröder, published in the paper of Klötzli and Grünig (1976) is worth mentioning. It seems, that the enrichment of water in nutritive materials to a certain degree is favourable to the development of reed plants and results in tall and thick shoots with abundant foliage. However, an extreme deterioration in water quality, an increase in its nutrient content may result in the decay of reed.

In Hungary there are some extensive lakes, the littoral zone or almost the whole surface of which are covered by reed stands. Among others such are Lake Balaton and Lake Velence. These both are recently threatened with eutrophication. The decay and the decrease in quality of previously unintermittent, closed stands of reed can be regarded as signs of this. It seems, that in Hungary only reed plants with a chromosome number of 48 are to be found.

It is rather surprising, that the great Hungarian botanist of the last century, Vince Borbás, in his book about the Lake Balaton, published in 1900, describes a condition of reed stands, which is very similar to that of present days. According to this, unintermittent stands grew at that time only on the northern, windless side of the lake. Whereas, on the southern, windward side, the reed in the water occurred only in scatterd patches. Also on the exposed sites of the Tihany-peninsula, it could be found only sporadically. Borbás explained the differences between the stands on the two sides by the effects of the wind and the visitors at the more frequented southern side.

From some later works it could be learned that the reed stands of Lake Balaton significantly expanded by the thirties. It is also known that in these times Lake Velence was covered by almost closed stands of reeds. This all suggests, that in accordance with our previous supposition, during the thirties the area of reed stands and the quality of reed in the mentioned lakes could increase in consequence of the enrichment of water in nutritives. The cutting open of reeds in Lake Velence some ten years ago and the regular dredging of the bottom of the lake could be, already from the beginning, injurious for the reed stands and additionally these latters got more exposed to the harmful effects of the wind and man as well. The proof of the fact, that similar mechanical effects befall the reeds of Lake Balaton is that unintermittent stands at the shore of this lake have also remained only at the sheltered, undisturbed sites.

But whether this would be the only cause of the decay of reed? Could also the decrease in reed quality be attributed to nothing but mechanical effects? These questions were to be answered by our research, carried out since 1985 at Lake Balaton as well as at Lake Velence. In the work of our research team not only botanists and ecologists but also plant physiologists, zoologists, mycologists, genetics, plant histologists, hydrologists and cartographers took part.

On the ground of our and of others' (Tóth, 1979; Gorzó, 1986) investigations it could be easily pointed out that, besides the listed changes, a number of alterations had come about in our lakes, which cannot be ascribed to mechanical effects. First of all, on many spots along the northern side of Lake Balaton and in Lake Velence as well a great quantity of organic matter has deposited in the reeds. This often forms a loose, 30-40 cm deep sediment. It is known of such deposits that they may result in permanent anaerobia and sometimes hydrogen sulphide accumulation, which is injurious to the underwater parts of reed.

In the loose sediment the rhizomes of the reed cannot take strong roots. Frequently some clumps of reed together with huge bundles of rhizomes are fastened down to the bottom below the sediment only by a single or a few perpendicular rhizomes. These few fastening rhizomes cannot keep up the large and heavy rhizome mass and the bundle breaks away from the bottom because of undulation. The least disintegration in such stands can lead to the total decay of them. In contrast, the reeds on a compact bottom and on the foreshore only seldom get sparse.

The data of the Water Conservancy of Middle Transdanubia show that the conductance of water has also changed. It reached its maximum value in 1983 and since that time slightly decreased. The calcium, the potassium and the hydrocarbonate ion content of the water of Lake Balaton have diminished during this period. The minimum values of calcium ion content coincided with the maximum trophicity of the lake. In the same period the amount of magnesium and sodium ions increased and reached its maximum value in 1984. The amount of chloride and sulphate ions has also been slowly increasing. On lake-shore sectors, where the loading by chloride is considerable, as for example at inflows of sewage water, first of all common cattail (Typha angustifolia) is spreading, and is occupying the place of reed. This species is capable of accumulating more sodium as well as chloride ions.

The composition of water microorganisms has also altered. Recently in summer or in late summer the nitrogen/phosphorus ratio decreases regularly under ten, what is favourable to the rapid multiplication of cyanobacteria. The extremely rapidly developing huge algal biomass bunches together on one or the other side - subject for the wind - in the littoral zone in the reeds and causes a firm sedimentation and precipitation of lime, increase in pH and anaerobia.

The biogen fall-out of lime is accompanied by a deprival of phosphorus and microelements as well. The tussocks of reed are often thickly covered by lime of biological origin, mixed with silt and algal biomass, which is hindering the shooting of the reed. This can rich such an extent, that only some underwater dead tussocks remained to show the former site of reeds. Hence the spring dominating diatoms characteristic for the 1970-s have been forced back and their biotope was occupied by the nitrogen-fixing cyanobacteria, dominating in late summer. And whilst diatoms are decomposed on the bottom of the lake, the same process by the floating cyanobacteria takes place in the littoral zone, depending on the direction of wind.

In the decaying reeds a green alga, Cladophora glomerata, can also be found in large quantities, which, creating a continuous cover on the surface of the shoots is also hindering the shooting and growth of young reed sprouts. The height of the reed stem therefore reaches sometimes only 80 to 100 cm-s. The presence of zebra mussel, Dreissena polymorpha, in large numbers on the underwater surface of not only dead but also living reed shoots is also troublesome.

The results of investigations of element content of reed did not help us too much in recognizing the causes of the decay of these populations of plants. On the basis of our Cluster analyses it can be concluded that the chemical element content of healthy and diseased reeds does not show significant differences when the single organisms (figure 2.), the organs of these organisms (figure 1.) or the elements contained by them are compared. This is also supported by comparing the samples from Lakes Balaton and Velence; the differences in the element content deriving from the geochemical environment were greater than the effects brought about in the same characteristics by the factors, causing the decline of reed. This all suggests that the decline of this plant species was neither caused by nutritional disorders, nor by heavy metals or acidification of the water. Some more significant differences between the healthy and diseased plants could be detected only in the rhizomes. It coincided to some extent with this observation, that a similarity between the K-content of the samples from the two lakes could be found also in the rhizomes and additionally in the roots as well (figure 3.). It is possible that the less K-content in the rhizomes of the diseased plants may be one of the causes of the decrease in the stability of

Cluster analysis

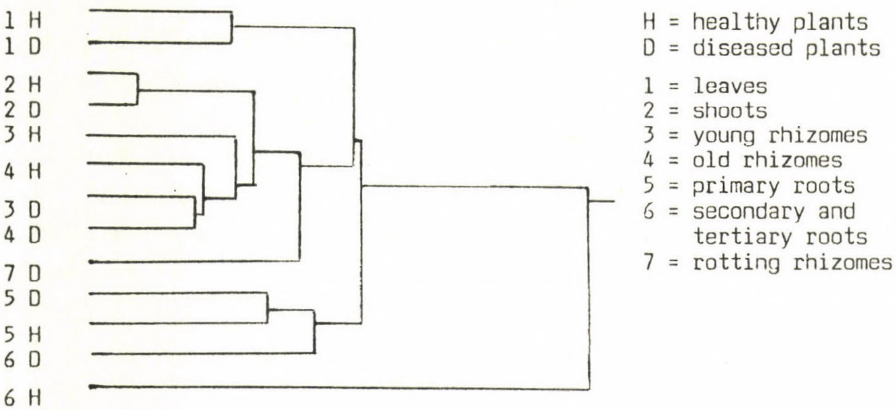


Figure 1. Grouping of the organs of reed on the basis of their chemical composition

Cluster analysis

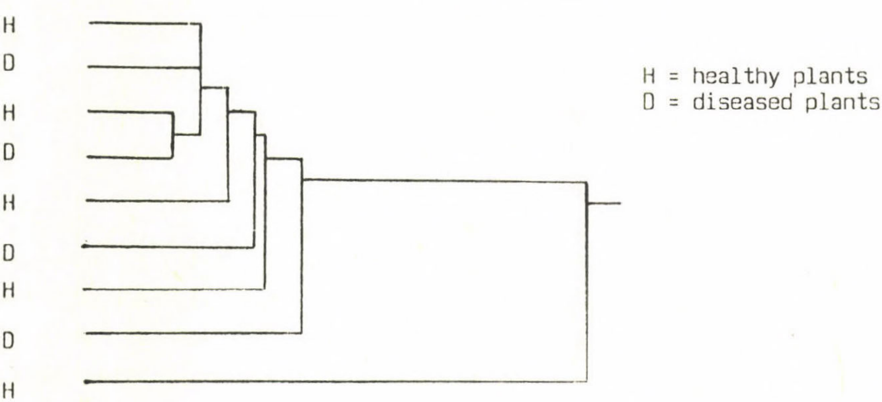


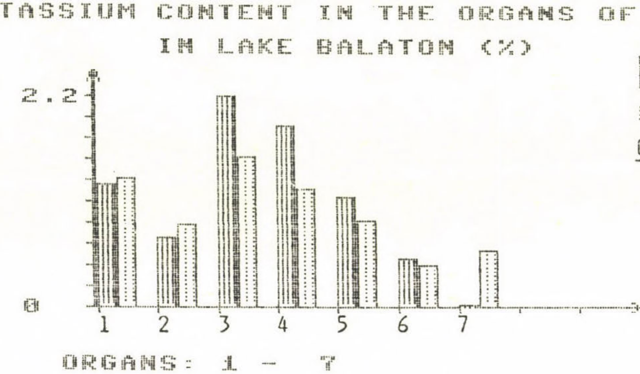
Figure 2. Grouping of healthy and diseased reed plants on the basis of the chemical composition of their young rhizomes

these plant organs; one of the most conspicuous morphological differences between the diseased and healthy plants is namely the presence of rotting rhizomes in the former ones. The decreased level of K in the underground organs of the reed may contribute to a decreased tolerance of parasites (fungi and insects). It was also conspicuous that in most organs of the healthy reed the N-content was higher than in the organs of the diseased plants (figure 4.).

Besides the above, the low molybdenum content (0.84 and 0.3 ppm in Lakes Balaton and Velence, respectively) of the decaying reed plants is also worth mentioning, however this value is rather low (1.0 and 0.54 in the two lakes, respectively) in the healthy plants too. Molybdenum deficiency may cause troubles in the nitrogen metabolism (see the leaves of the reeds in Lake Velence!), the most characteristic consequence of which can be the low chlorophyll content in the leaves.

The rhizomes of the perennial reed plant, reproducing itself by generative means rather poorly, have an enormously important role in the reproduction. Their lifetime is between 3 and 7 years. In spring the transport of nutrients from the rhizome makes possible the plant to shoot, whilst in autumn the reserve materials are stored also in its tissues. There are white and young, older and yellowish-brown as well as rotting rhizome internodes distinguished. The latest ones occur first of all in the decaying plants. In the senescing and rotting rhizome segments the amount of nitrogen, potassium, phosphorus and chlorine is decreasing, whereas the proportion of calcium, magnesium, aluminum, barium, stroncium and some other microelements is increasing. These results are in accordance with those experienced in withering leaves.

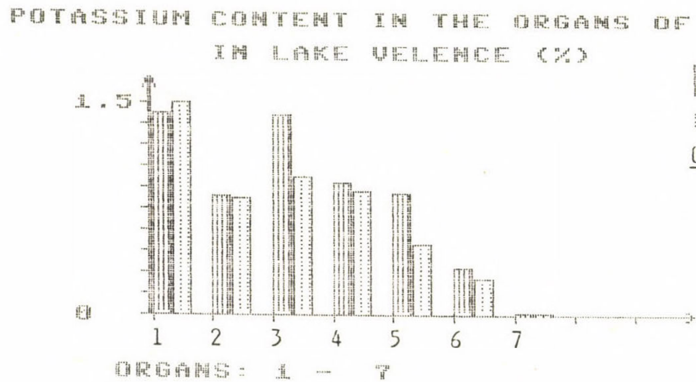
In the decaying reed plants rotting rhizome internodes can be found - as already mentioned - frequently. In these iternodes the amounts of volatile fatty acids - such as acetic acid, propionic acid, butyric acid, valeric acid, caproic acid - were also analysed (table 1.). It was verified that the values of fatty acid content were very high. In some cases the total amount of fatty acids was as high as 268 ppm. Propionic acid (its maximum value was 21 ppm) and acetic acid (its maximum value was 55 ppm) are regarded as the most dangerous of all components. The acumulation of these and other fatty acids may be the consequence of decomposition processes taking place under anaerobic conditions. The accumulation of fatty acids may have a bioindicative role in the prediction of endangerment of reed stands.



H - healthy
D - diseased

<u>Organs</u>

1 - leaves
2 - shoots
3 - young (white) rhizomes
4 - old (brown) rhizomes
5 - primary roots
6 - secondary and tertiary roots
7 - rotting rhizomes

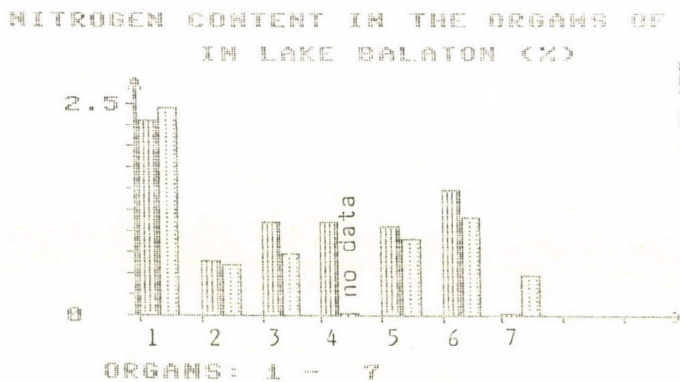


H - healthy
D - diseased

<u>Organs</u>

1-leaves
2- shoots
3 - young (white) rhizomes
4 - old (brown) rhizomes
5 - primary roots
6 - secondary and tertiary roots
7 - rotting rhizomes

Figure 3. Potassium content of reed plants



H - healthy
D - diseased

Organs

1 - leaves
2 - shoots
3 - young (white) rhizomes
4 - old (brown) rhizomes
5 - primary roots
6 - secondary and tertiary roots
7 - rotting rhizomes

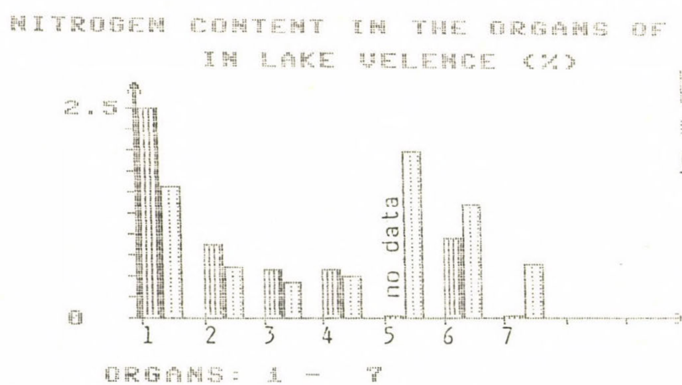


Figure 4. Nitrogen content of reed plants

H - healthy
D - diseased

Organs

1 - leaves
2 - shoots
3 - young (white) rhizomes
4 - old (brown) rhizomes
5 - primary roots
6 - secondary and tertiary roots
7 - rotting rhizomes

Table 1. Volatile fatty acid content (in ppm) of the rhizomes of reed in Lake Balaton

Volatile fatty acids	Rotting rhizomes of decaying reeds	Rhizomes of healthy reeds
acetic acid	4.1- 55.2	9.4-12.6
propionic acid	0.7- 21.1	0 - 1.0
l-butyric acid	0 - 1.7	0
n-butyric acid	0 - 94.9	0 - 1.6
l-valeric acid	0 - 3.7	0
n-valeric acid	0 - 12.0	0
caproic acid	0 -100.1	0 - 0.5
altogether	5.5-268.2	9.4-14.7

The results of the morphological and anatomical investigations show that the rhizome internodes of diseased stands are shorter and thinner than those of healthy stands. Due to the dense arrangement of internodes the root system becomes more web-like. The spots of injuries, which can originate from mechanical ruptures or attack of pathogens, are indicated by brown chromatism as a result of suberification. These injuries will very likely take place more easily, if the nitrogen content of the reed is high. If the injury reaches such an extent, that the central rhexigen cavity is also included, the danger of infection increases and a total desorganization of the whole internode may occur. This is indicated by the presence of bacterium cultures and fungal tissues as well as by the proliferation of cells. If the nodes are not injured, they can act as fairly effective filters against pathogens and their toxic materials. The elimination of diseased internodes interrupts the continuity of the rhizome. As a consequence the developing shoots get less nutritives and the dormant buds of rhizome sections, remained without any shoot, begin to grow forcedly. In both cases the result is the formation of thin, stunted shoots. If the rhizomes consist of numerous such diseased, eliminated internodes, this phenomenon can be one of the causes of decaying of reeds.

Insects, invading the embrionary apices of the young shoots cause drastic changes in the upper internodes. That part of the stem, in which their larvae develop, is covered by the shield of leaf-sheats. The leaf blades practically cement together with the apical desorganized tissues, thus providing a larval nest. It is probable that in such cases the majority of nutrients will be transported into the shoot apex, to promote the feeding of the insect and the forming of protecting tissues which produce the gall. This process may reduce the lignification in the lower internodes and the stem may remain weak. Death of the apical bud and changes in the distribution of hormones may lead to the development of lateral buds. New lateral shoots may sometimes appear also on irregular points of the stem. As a consequence of the increased lateral shoot formation, densely branched reeds of poor quality develop.

According to our physiological investigations the chlorophyll and the total pigment contents in the withering leaves of decaying reeds are lower than in the healthy plants. This can lead to a decreased photosynthetic performance and productivity. The indicative value of these changes can be also used for the prediction of endangerment of stands. These pigment content investigations however unfortunately cannot exclude any of the possible causing factors of reed decay, since the pigment structure of higher plants can respond only by similar changes to the most different outer and inner injurious effects.

In the rhizomes of decaying reed stands less soluble sugars and starch accumulate than in the rhizomes of healthy stands. In the leaves of decaying reeds however more starch could be found than in the controls. Such a distribution of carbohydrates suggests, that under the influence of some sort of stress the transport of triosephosphates from chloroplasts is interrupted.

Different pests and plant parasites appear on reed in enormous quantities. This is a general feature of weakened stands. 19 invertebrate pests have been so far identified in Lake Balaton and Lake Velence, among which there were thrips, plant hoppers, plant-lice, bugs, beetles, moths, flies and mites. These pests injure the most different organs (rhizomes, stem, leaves, inflorescence, fruit) of reed, weakening its stem, hindering fruit set and allowing the establishment of other parasites.

It is remarkable that the harms caused by mealy plum aphid occurred first of all along the water-side of the stands, where the decay of reed is most extensive. We consider the harms caused by some beetles, as for example Donacia spp., Plateomaris braccata, as very serious. These take up oxygen from the aerenchyma of the rhizome by their pointed chitin tubes located on the tip of their abdomen. After drawing out the chitin tube, some water enters the internode of the rhizome. Most of rhizome internodes having the wounds of such origin have begun rotting. The species mentioned occur first of all in decaying reeds.

The role of Cladophora and Dreissena polymorpha, which are found in some stands in great quantities, is still to be cleared up. Schröder (1987) suggests that Cladophora if decomposing under anaerobic conditions may be toxic to reed.

Two Puccinia (Puccinia phragmites, P. magnusiana) and one Ustilago (U. grandis) species of microscopic fungi are considered to be most dangerous to the reed.

On the basis of a manyfold approach we conclude that the reeds of Lake Balaton and Lake Velence are in a rather bad condition because of several causes. Our proposal for the measures to be taken was first of all a winter removal of the dead shoots from the lakes. This would decrease eutrophication in the littoral zone and would additionally decrease the survival of parasites in the winter period. However, the role of man in the eutrophication and mechanical disturbances also cannot be neglected.

LITERATURE

Borbás, V. (1900): A Balaton tavának és partmellékének növényföldrajza és edényes növényzete. Budapest

Gorzó, G. (1986): A vízminőség változása a parti sávban. Manuscript 12 pp.

Klötzli, F. - Grünig, A. (1976): Seeufervegetation als Bioindikator. Daten und Dokumente zum Umweltschutz 19: 109-131.

Schröder, R. (1987): Das Schilfsterben am Bodensee-Untersee. Beobachtungen, Untersuchungen und Gegenmassnahmen. Arch. Hydrobiol. Suppl. 76: 53-99.

Tóth, L. (1979): A Balaton vízminősége. In: Baranyi, S. (ed.): A Balaton kutatási eredményeinek összefoglalása. Vízügyi Műszaki Gazdasági Tájékoztató 112: 220-261.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

EXPLOITATION AND MANAGEMENT OF FISHERY RESOURCES IN LAKE BALATON

P. BIRÓ

Balaton Limnological Research Institute of the
Hungarian Academy of Sciences
H-8237 Tihany, Hungary

INTRODUCTION

Lake Balaton is a shallow (average depth is 3.3 m) eutrophic lake of 595 km^2 in the western part of Hungary. Relics from early settlements indicate that Lake Balaton has been fished by man since 5000 B.P. (Sági, 1974). From that time there were basic changes both in methods used and in abundance of fish stocks (Biró, 1977). Flood control projects in the 19th century further affected the fish stocks and the fishery throughout water level fluctuations and elimination of important marshy areas i.e. spawning grounds (Tölg, 1963). Fishermen at Lake Balaton had since long formed professional groups because the handling of large seine nets required several people. We have records about the existence of fishing associations from the eighteenth century. Effective regulations were introduced in 1884 with the establishment of the Balaton Co-operatives (Répássy, 1909). Subsequently, the Balaton Fishing Corporation, founded in 1900, rented the entire lake for fishing for 50 yr. The Corporation improved the fishery through stocking, collection of fishery data, protection of spawning pikeperch and modernization of seines and technics of fisheries (Lukács, 1932). Ice fishing looked back to long tradition, however, it was prohibited in 1957. In 1948 the fisheries of Lake Balaton were taken over by the state (Woynárovich, 1958). Since 1975 the management of fisheries came under the authority of the Ministry of Food and Agriculture.

FISH FAUNA OF THE LAKE

47 of the 74 fish species inhabiting the Carpathian Basin are known in our lake and its drainage area. 13 nonnative species were introduced since the last century, or some of them appeared by chance. 20-24 species are quite frequent, but the fish capture generally contains only of 15-17 species. Pikeperch (Stizostedion lucioperca), carp (Cyprinus carpio)(restocked for anglers), bream (Abramis brama), razor fish (Pelecus cultratus), asp (Aspius aspius), pike (Esox lucius), sheatfish (Silurus glanis) etc. are of crucial importance. Eel (Anguilla anguilla) was introduced in 1961 and silver carp (Hypophthalmichthys molitrix) in 1972 in order to increase the fish production of the lake, which (commercial yield) varies between 7.6 and 46.3 kg ha^{-1} along the longitudinal trophic gradient of Lake Balaton from NE to SW (Biró and Vörös, MS).

Fish populations in Lake Balaton have undergone changes from the individual or combined effects of eutrophication, overfishing (commercial + sport), interspecific competition from native and nonnative species, and loading of nutrients and toxic compounds, as well as different environmental perturbations and human impacts (Biró, 1977). Some endangered and formerly believed to be extinct fish species (e.g. Perca fluviatilis, Micropterus salmoides) disappeared from the lake itself and at present they have self-sustaining stocks in different rivulets, as refugium areas (Zalewski et al., MS).

DYNAMICS AND EXPLOITATION OF FISH STOCKS IN LAKE BALATON

Trophic relationships and dynamics of various fish species occurring in great numbers have been studied in detail. Selected parameters for bream (Abramis brama) and pikeperch (Stizostedion lucioperca) have been presented in Table 1. All of these data are of crucial importance in understanding the responses of fish populations to biotic and abiotic effects.

Table 1. Selected population parameters for bream (for the years of 1973 and 1983) and for pikeperch (for the years of 1973 and 1984)

		Bream		Pike-perch	
B	kg/ha	160	180	9.7	1.6-4.9
N	i/ha	270	300	15	2.4-7.5
P	kg/ha	44	49	4.9	0.9-2.6
$P/\bar{B}$	%	72.6	?	50	39-63
Y	t/yr	1004	543	119	45.2
	kg/ha	16.8	7-42	2	0.8-2.8
A	%	62	?	65	30-71
F		33		67	25-96
M		64		37	11-34
S	%	38		35	29-70
$\bar{L}$	cm	19	19	36	35-43
t_{min}	yr	2.4		2.9	3-3.6
t_{max}	yr	9-13		10-12	10-13
E	%	41		41	45

Biomass (B), populations density (N) and production (P) increased in bream, but decreased in pikeperch. The $P/\bar{B}$ ratio practically remained the same, but yield (Y) in both species decreased significantly. Annual (A), fishing (F) and natural mortality (M) did not change significantly, however, they are widely variable in space and time. Rate of survival (S) probably increased in both species. Average length ($\bar{L}$) of fish caught practically remained the same and also the minimum and maximum ages at first capture remained stable (t_{min} and t_{max}). No considerable alteration of rate of exploitation (E) was observed during the period of 1973 and 1984.

Commercial fisheries in Lake Balaton are today performed by the Balaton Fishing Company. Five fishery stations (Siófok, Tihany, Balatonszemes, Fonyód, Keszthely) (Fig. 1) are situated around the lake, originally with 15 fishermen each, 2 towing ships and one 1000 m long, 5 m deep seine net of 4 cm cod-end-mesh size.

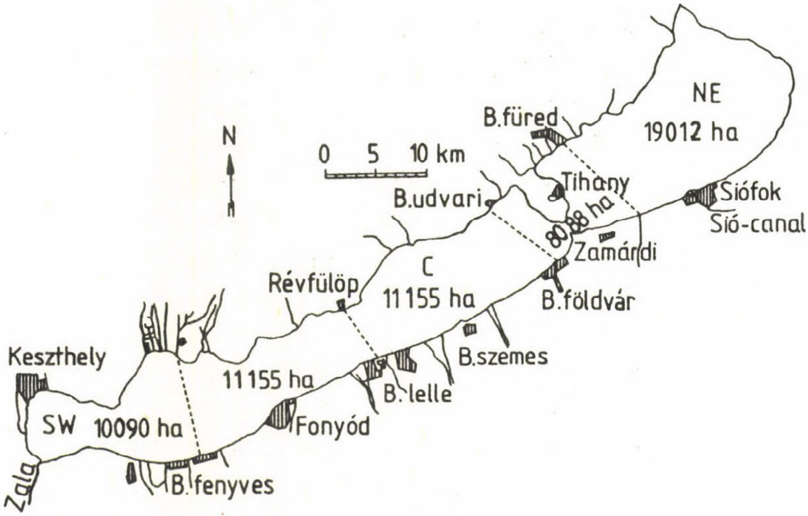


Figure 1. <u>Position of fishery stations and fishing areas</u>

Since the 1950s an increasing demand for sport fisheries has been registered. Now the estimated (enormously increased) number of sport fishermen varies somewhere between 100 000 and 200 000. Because of their demand the commercial fisheries were regulated in space and time and in quotas since 1978. It means that the maximum amount of fish harvested annually is fixed in 1200 t, except the introduced eel and silver carp which species can be recaptured without any limitation. Commercial fishing has been prohibited in the littoral zone within 100 m along the northern shoreline and within 200 m along the southern one.

Due to the fluctuations taken place in the stocks of different native species mainly in consequence of environmental perturbations etc., the fishing right became a crucial debate between commercial and recreational fisheries. During the last couple of years there were many unacceptable "explanations" for the declines of fish stocks blaiming the commercial fisheries, but excluding all the scientific observations and especially the basic and rapid changes in water quality, as well as long lasting effects of earlier massive fish kills (1965, 1975, 1982).

Recorded commercial landings of various fish at several areas were quite different and varied with the annual fishing effort for a long run (Fig. 2). Bream followed by pikeperch are caught in greatest quantities all the time. Bream represents the overwhelming majority of the annual catch (about 70-80 %). Pikeperch amounted to 6-12 % (1-3 kg ha^{-1}). Figure 3 shows the variation of pikeperch catches by commercial and sport fisheries. The commercial fishing effort decreased during the last two decades and commercial pikeperch catches dropped down to about 1/3 of their former values between the 1960s and 1980s, and those of sport fishery have increased roughly about ten times.

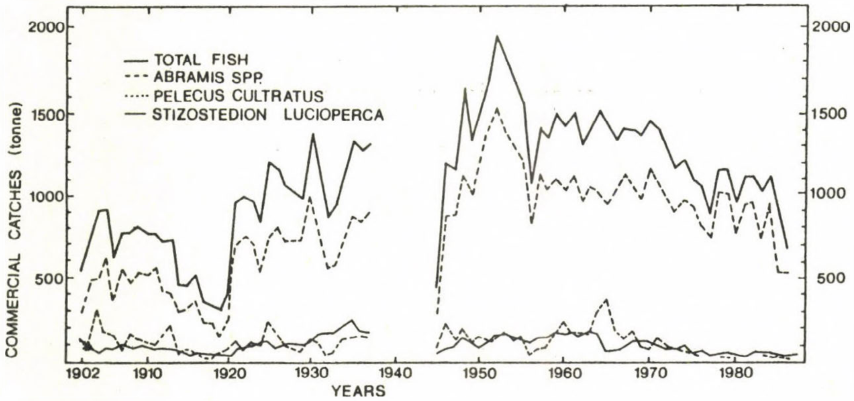


Figure 2. Commercial fish catches in Lake Balaton, 1902-87.

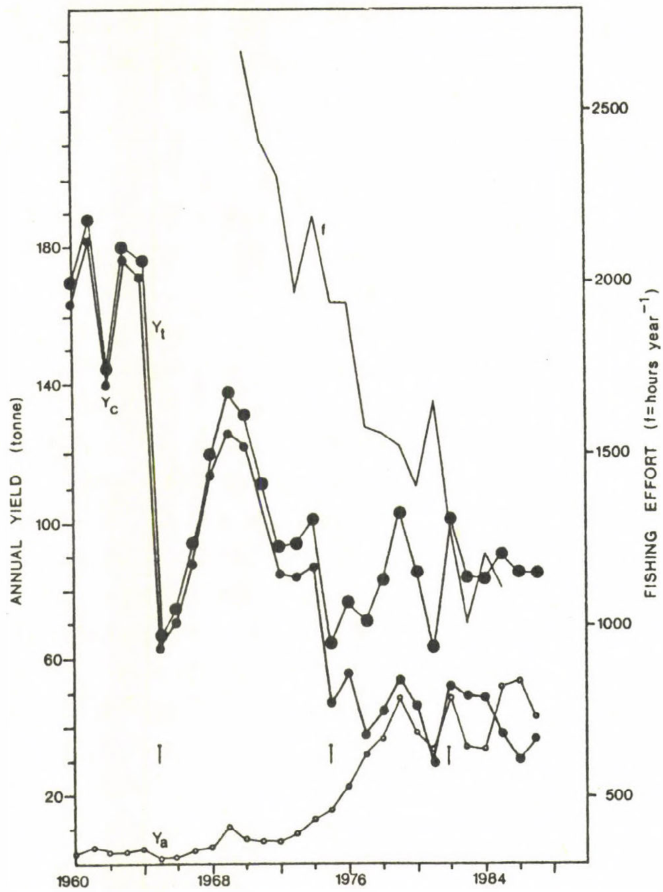


Figure 3. Pikeperch catches in Lake Balaton, 1960-87.
(Y_t = total yield, Y_c = commercial harvest, Y_a = angling yield, f = annual fishing effort, arrows indicate the years of massive fish kills)

The same increase was registered for the total annual sport fish harvest.

Number of fishing licences sold between 1977 and 1986 increased from 44 203 to 100 197 according to the statistics of Hungarian National Angling Association (MOHOSZ).

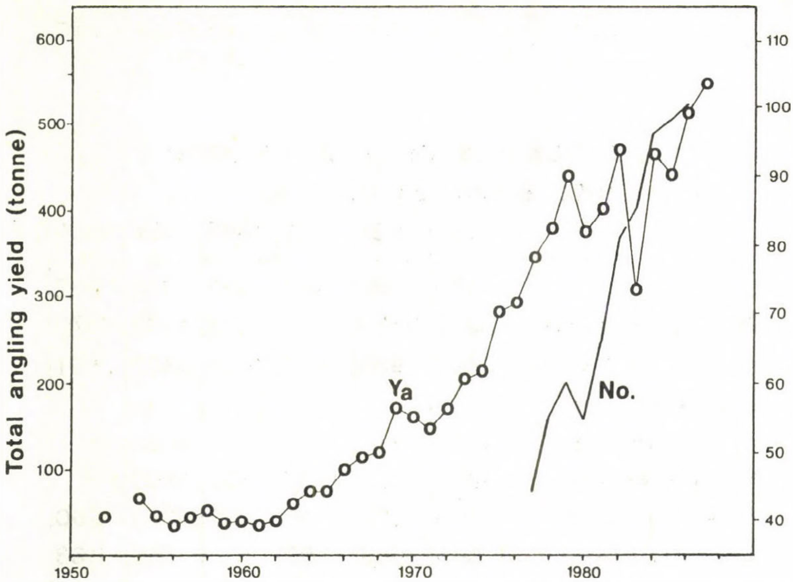


Figure 4. Increase of angling yield in Lake Balaton, 1952-87.

Because of wide oscillation in the yields (and probably in the productivity, as well as in the stock density) of different fish species, just all of them can be maintained at nearly the same level with regular stockings (Table 2). Carp has been stocked for angling purposes with two-summer old material, because its natural spawning is unsuccessful. This species inhabits the littoral zone and is not the object of commercial fisheries. Predatory fish, such are pikeperch, pike and asp are stocked usually in fry stage. Placing of nests promotes the spawning success of pikeperch and following the protected hatching of eggs, larvae are released to the lake. Not only the amount of stocked fish increased more or less, but also the costs of stockings showed an increasing tendency (Table 3).

Table 2. Annual fish stockings in Lake Balaton, 1980-85.
(maximum sizes of fry are indicated)

FISH STOCKINGS IN LAKE BALATON
(pc x 1000, except as indicated)

	1980	1981	1982	1983	1984	1985
Carp (t)	170	180	182	223	226	232
Pikeperch nests	6273	5883	5250	5576	4693	2570
Pikeperch fry 4cm	992	935	912	1520	3387	2521
Pikeperch O+ 11-15 cm	4400	–	–	–	8420	–
Eel larvae	3200	4000	1900	2300	5800	–
Pike fry 3cm	100	200	200	250	250	250
Asp fry 4cm	585	1885	50	–	168	1163
Silver carp 1+ (t)	42	40	40	20	–	–
Bream nests		15700	17656	8141	10683	11414

Table 3. Annual costs of fish stockings in Lake Balaton

COSTS OF FISH STOCKINGS (x1000 Ft)

	1983	1984	1985	1986
Total costs	15 259	18 929	22 044	26 948
Carp	10 118	11 003	15 219	17 058
Pikeperch	2 976	4 869	2 434	5 632
Eel	1 700	1 728	–	3 540
Others*	1 465	1 329	4 391	718
Angling licences	16 067	17 325	18 189	18 417

* Pike, Asp, etc.

MANAGEMENT MEASURES

Fish population data show great variability between perceived effects and apparent causes and require a long time to determine true causes of perceived effects. Such circumstances make it clear that a simple "trial and error" approach to managing a fishery is usually a poor one. Fishery biologists are increasingly confronted with situations involving conflicts between competing uses of aquatic resources, where quick decisions are required (Everhart and Youngs, 1981).

Sport fishing is one of the most popular forms of outdoor recreation. While the number of anglers in Hungary is increasing many times faster than the population, the total area of fishing waters is at best static. The problem for both commercial and sport fisheries is to find ways to maintain and improve aquatic environments, because of increased exploitation of the same fishery resources.

Fish stocks in Lake Balaton are intensively fished for sport and commercial fisheries. Some attempts have been made to manage fisheries by quota, minimum legal size and minimum mesh size. The managers of our fishery have five basic options to control the fishery. These are gear control (selectivity with mesh size regulation), season control (closed seasons during spawning period), limited entry, pricing the right to fish (licenses) and quotas (maximized harvest). Control of fishing gear, for example stipulating the mesh size to be used, is intended to permit sufficient escapement of young fish so that they may later be recruited to the adult population and sustain the stocks nearly, or at least at the same level (Craig, 1987). They are also used to control fishing effort, for example, in the number of nets set per day, or in annual fishing efforts, or in the number of rods used per anglers and their daily quotas per capita. Closed seasons (in case of all valuable species), closed areas (for commercial and, in a few cases, for recreational fisheries) and shortened seasons aim to protect fish during critical periods in their life cycle such as spawning periods and also to control fishing effort. However, these measures only reduce the time in which the fish may be harvested and usually increase the fishing effort (in our case both commercial and sport fish harvest).

Limiting entry to the fishery may be a viable management option (Craig, 1987). Experience has shown that the use of quotas may be the best method of controlling the level of harvest and also in controlling investments commensurates with the individual production. The removal of predators (or their massive kills from outer causes) often result in an increase in smaller forage fish or less valuable cyprinids (e.g. Carassius auratus gibelio). Alternatively a sharp increase in predatory fish causes a decrease in overal yields which has been demonstrated in a number of Polish lakes (Bonar, 1977). Restrictions on size limits are intended to maximise the yield, increase the catch of larger fish and protect potential spawners. However, a trial of increase in size limits (in pikeperch from 32 to 40 cm standard length) and mesh size during the middle of 1970s resulted in a decrease in the yield of large fish, and an increase in the yield and number of young and smaller sized fish.

Stocking is a common practice and can be of three types (Craig, 1987). The first type is introductory (eel, silver carp), when the species stocked is new in the habitat. The second type is maintenance stocking where the fish present do not reproduce naturally or if they do it is very limited (common carp). The third type is supplementary stocking to augment natural reproduction of existing fish (the majority of fish species inhabiting Lake Balaton). These stockings were often the basis for "put and take" sport fisheries (as it is in the case of carp). The cost of stocking must also be related to income from the fishery, however, the expenses are always increasing because of increasing price of planting material.

Human impacts influence the abundance and quality of different fish species of Lake Balaton. Many fish communities and stocks are depressed (pikeperch, asp, pike, bleak), collapsed (Pelecus cultratus), or extinct (Gobio gobio), or emigrated (Perca fluviatilis) as a result of severe stresses brought about directly or indirectly by man (eutrophication, loading of nutrients and toxic compounds, introductions etc). To prevent the lake biota from further alterations, the most important step is to restore the mesotrophic water quality. The restructuring of inshore areas and shoreline and the re-annexation of marshy areas would promote a more healthy functioning of the ecosystem (Biró, 1977). Several factors need further study. These include a more exact description of the pathways of material and energy flow (Biró and Vörös, 1988), as well as of the alterations in the stability, diversity and resilience of the ecosystem under human impacts, the development of an optimal and plastic strategy for commercial and recreational fisheries in relation to the lake's real production capacity, the detailed exploration of processes taking place at various trophic levels and a quantitative description of their interaction at individual, population and community levels.

Acknowledgements

I wish to thank L. Elek (Balaton Fishing Company, Siófok) and J. Páskándy (Hungarian National Angling Association, Budapest) for fisheries statistics in Lake Balaton. This work was financed by AKA Programme ("Akadémiai Kutatási Alap", theme no. 30054).

REFERENCES

Biró, P. (1977): Effects of exploitation, introductions and eutrophication on percids in Lake Balaton. J. Fish. Res. Board Can. 34: 1678-1683.

Biró, P. and L. Vörös (1988): Relationships between the yield of bream, Abramis brama L., chlorophyll-a concentration and shore-length:water area ratio in Lake Balaton, Hungary. Aquaculture and Fisheries Management 19: 53-61.

Biró, P. and L. Vörös (MS): Trophic relationships between primary producers and fish yields in Lake Balaton. (15 p. + 2 tables + 7 figures)

Bonar, A. (1977): Relations between exploitation, yield and community structure in Polish pikeperch (Stizostedion lucioperca) lakes, 1966-71. J. Fish. Res. Board Can. 34: 1576-1580.

Craig, F. J. (1987): The Biology of Perch and related species. Croom Helm Ltd., Provident House, Burrell Row, Beckenham, Kent. 333 p.

Everhart, W. H. and W.D. Youngs (1981): Principles of Fishery Science. 2nd ed., Comstock Publishing Associates, a division of Cornell University Press, Ithaca and London. 349 p.

Lukács, K. (1932): A Balaton halai gyakoriságáról. Magyar Biol. Kut. Munk. 5: 17-27.

Répássy, M. (1909): Édesvizi halászat és halgazdaság. M.Kir. Földmivelésügyi Miniszter (ed.) No. 4, "Pallas" R.T., Budapest. 502 p.

Sági, K. (1974): A Balaton-vidék történelme és régészeti emlékei. pp. 337-356. In: Balaton monográfia. K. Tóth (ed.) Panoráma, Budapest.

Tölg, I. (1963): A hajdani berkek lecsapolásának hatása a Balaton mai halállományának minőségére. Hidrológiai Közlöny 43: 77-81.

Woynárovich, E. (1958): A balatoni halgazdálkodás jelentősége. Földrajzi Közlemények 6(82): 389-392.

Zalewski, M., P. Biró, M. Przybilski, I. Tátrai and P. Frankiewicz (MS): The structure of fish communities in streams of north part of the catchment area of Lake Balaton. (12 p. + 2 tables + 2 figures)

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest
Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

DIATOMS FROM BOTTOM SEDIMENTS OF LAKE BALATON AND THEIR ANALYSIS WITH THE PROGRAMS DECORANA AND TWINSPAN

E. KŐVÁRI-GULYÁS

Consulting Engineering
H-1088 Budapest, Reviczky u. 4, Hungary

INTRODUCTION

In the early 1980s a geological program was started in Hungary. The aim of the project was to observe recent changes of the environment and to evaluate past geological events on the basis of these data. In the framework of the project the large, shallow Lake Balaton was studied, too. Bottom sediment core samples were collected and analysed paleontologically, sedimentologically, etc. The project is to be finished in the 1990s by a general evaluation of the data.

In this paper the diatom content of four boreholes is described in the Eastern Basin of Lake Balaton. Several algological studies have been undertaken in Lake Balaton, including some studies on the recent diatoms of the bottom sediments (1,2). This paper is based on the Decorana and Twinspan computer programs applied to subfossil diatom assemblages in Hungary.

MATERIAL AND METHODS

The location and depth of the boreholes are shown in Fig. 1. The sediment was Holocene silt covered by 20 cm and 1 m Upper-Pannonian gray silt at stations 9, 11 and 15, 16, respectively. The age of the layers was determined palynologically.

The core samples were cut into 20 cm layers. Quite a lot of subsamples were free of diatoms. Thirty-two subsamples containing diatoms were included in the multivariate analysis.

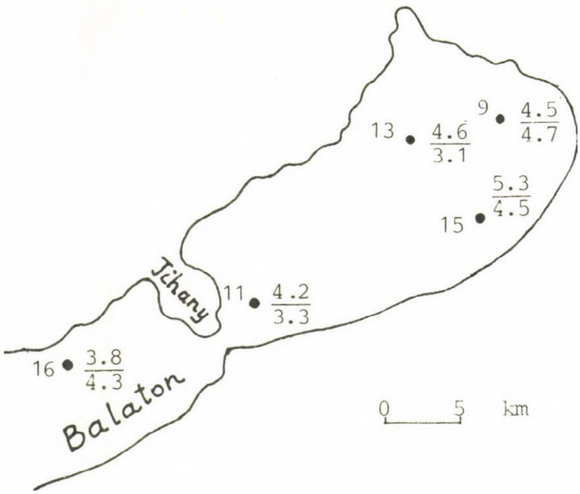


Fig. 1. Borehole location and depth.
Explanation: ● 11 = site number; $\frac{4.2}{3.3} = \frac{\text{water depth}}{\text{core length}}$

Three slides were prepared from each subsample in the traditional way.

Diatoms were studied by light and scanning electron microscopes. The nomenclature used is in accordance with the Van Landingham's catalogues (3). Six hundred specimens per slide were identified and counted for the Decorana and Twinspan multivariate analysis.

The Detrended Correspondence Analysis (Decorana) and the improved variant of the previous Reciprocal Averaging (RA) method give a spectacular ordination of the samples based on their diatom contents. The Twinspan method allows an interpretation of the species ordination.

The Two-way Indicator Species Analysis (Twinspan) is a polythetic divisive method of classification. Its most significant new feature is that the samples are first classified by repeated dichotomization and then the results are used to generate a species classification. The final results appear in an ordered two-way table showing the types and quantitative relations of the classifications. This method improves the paleoecological interpretation of the data.

OBSERVATIONS

Of the 127 identified diatom species, 20 have not been found previously in Lake Balaton. Two species, Amphora ajajensis Skab. and Campylodiscus lacusbaikali Skvortz. are new species in Hungary.

Station 9 can be characterized by the dominance of Epithemia zebra (Ehr.) Kütz., Fragillaria pinnata Ehr., Melosira granulata (Ehr.) Ralfs in the deeper layers. At Station 11 Cocconeis diminuata Pant., Diploneis elliptica (Kütz.) Cl. and Navicula scutelloides W.Sm. were dominant at the top, whereas Melosira granulata (Ehr.) Ralfs, Navicula oblonga (Kütz.) Kütz., Melosira ambigua (Grun.) O. Müller and Anomoneis costata (Kütz.) Hust. in the lower layers. Fragillaria pinnata Ehr., Fragillaria construens (Ehr.) Grun. and Cymbella ehrenbergii Kütz. var. hungarica Pant. occurred in the greatest number in the upper part of the core from station 15.

Only a few or no diatoms could be found in any of these cores in the layers 2 m or more from the top of the sediments. Such a sharp change of the diatom assemblage was not observed at station 16. This core contained diatoms in variable numbers along its whole length. Near the sediment surface only a few diatoms occurred, whereas in the deeper layers Fragillaria pinnata Ehr. and Cyclotella comta (Ehr.) Kütz. were dominant. Between 1.8 and 3.6 m the diatoms were less abundant, but Campylodiscus noricus Ehr. var. hibernicus (Ehr.) Grun. appeared. This species together with Melosira ambigua (Grun.) O. Müller and Rhopalodia gibba (Ehr.) O. Müller was characteristic in the deepest layers.

On the Decorana site diagrams station 16 was separated from the other stations along axes 1 and 2 as well as along axes 2 and 3. At the second level distribution of the Twinspan diagram the deepest layers of stations 9 and 16 were grouped into one cluster. The Decorana and Twinspan species diagrams did not show significant or discernible changes in the diatom assemblages.

DISCUSSION

The diatom analysis at the four stations was a part of a new geological program aiming at the better understanding of the paleoenvironmental changes in Lake Balaton.

The absence of diatoms in layers 2 m under the surface of the sediments may be related to the sulphur precipitation observed in these layers. The separation of station 16 on the Decorana diagram may be explained by the morphology of the basin, though further data are needed from other stations along the western shore of the Tihany peninsula in order to verify this hypothesis. More evidence is necessary to support the hypothesis that the Twinspan cluster of the deepest layers has a chronological basis. In the western basin of Lake Balaton (Keszthely area) Quaternary sediments occur together with Holocene and Upper-Pannonian layers.

Further diatom analysis and computerized evaluation related to chemical data open the possibility of paleoenvironmental reconstruction as well as a better understanding of the history of Lake Balaton.

ACKNOWLEDGEMENTS

I am grateful to my colleagues in the Hungarian Geological Institute, the Balaton Limnological Research Institute, the British Museum, the London University and the Freshwater Biological Association, Ambleside, who helped me during the research.

REFERENCES

1. Pantocsek, J. (1902). Die Bacillarien des Balatonsees. 1-112, Wien.
2. Szemes, G. (1957). A Balaton kovamoszatai. Ann. Inst. Biol. Tihany, 193-270.
3. Van Landingham, S.L. (1967-1978). Catalogue of the fossil and recent genera and species of diatoms and their synonyms. I-VI, 1-3605. Lehre-Vaduz.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

POST-PROJECT ANALYSIS OF THE RECONSTRUCTED KIS-BALATON

A. DIÓSI

Ministry for Environment and Water Management
H-1011 Budapest, Fő utca 44-50, Hungary

HISTORICAL BACKGROUND

Up till the middle of the 19th century, the Kis-Balaton (meaning Small Balaton) was part, in fact, the most western basin of Lake Balaton. However, during the hundred years from 1837 to 1937, great changes have taken place in that large, but very shallow, partly marshy area of the lake.

The slow ageing of the original Kis-Balaton was hastened by human interference. An earth embankment with a bridge constructed between Kis-Balaton and the main lake, and the lowering of the main lake's water level first reduced, then blocked the backflow of Lake Balaton water to Kis-Balaton. By that the two-directional exchange of water between the two parts of the originally unified lake was brought to an end, and Kis-Balaton became a lake of its own. (The contours and position of the old Kis-Balaton are shown in Fig. 1.)

Following all that, the largest part of Kis-Balaton was drained and turned to agricultural use. Yet, a low lying 14 km^2 wetland full with aquatic plants and animals was left undisturbed. This area has become a known bird sanctuary, designated also as important waterfowl habitat by the Ramsar Convention on Migrating Birds.

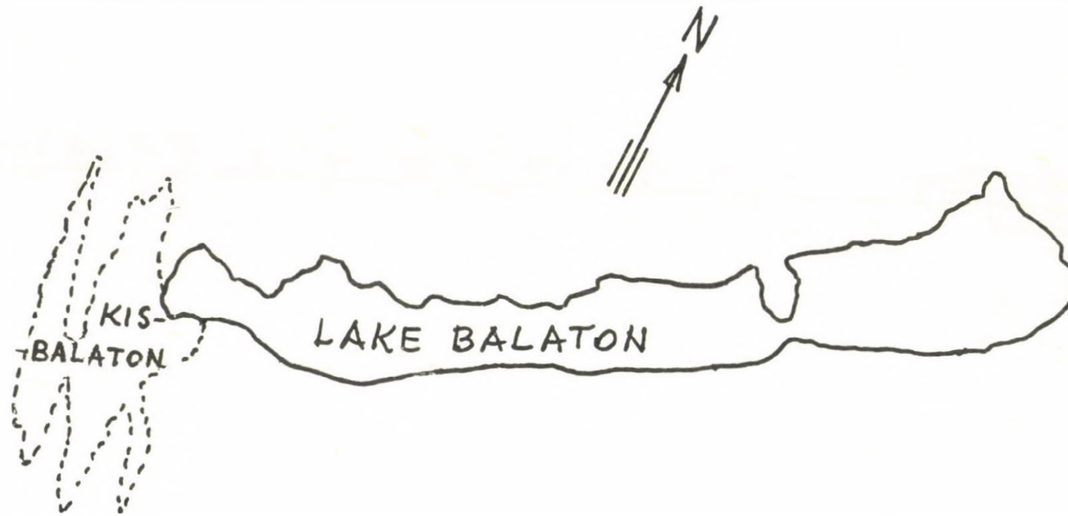


Figure 1. Layout of Lake Balaton and Kis-Balaton

RECONSTRUCTION OF THE ORIGINAL KIS-BALATON

Due to the drying up of most part of the old Kis-Balaton, the water purifying function of the same was transferred to the then most western part of Lake Balaton, the Keszthely Basin.

Unfortunately, during the same period of time, and particularly from the 1960's onwards the man-made pollution of River Zala, the main tributary of Lake Balaton, and consequently that of the Keszthely Basin has rapidly increased.
Under these circumstances, the existence of the Kis-Balaton lake/wetland system could have been more useful than ever before.

Meanwhile the agricultural productivity of the drained land fell short of expectations, making the idea of reconstructing the old Kis-Balaton a reasonable suggestion.

Decision about the matter was made in March, 1983, and the construction work of Phase I - which comprises the smaller sized upper reservoir (Reservoir I) of the proposed complete system of two, more or less independent lakes - also started in the same year. (Main features of Lake Balaton and the Kis-Balaton Reservoirs, the total for Reservoir I and Reservoir II are given in Table 1.)

Table 1. Main features of Lake Balaton and Kis-Balaton

	Lake Balaton	Kis-Balaton Reservoirs
Catchment area	5776 km^2	2622 km^2
Surface area	600 km^2	69.4 km^2
Length	78 km	
Width	7.7 km	
Volume	1860 mil. m^3	85 mil. m^3
Average depth	3.1 m	1.2 m
Mean flow (all inflows)	18 m^3/s	9 m^3/s
Retention time	3.3 y	110 d

Phase I, that is Reservoir I, was completed by June, 1985, and since that time it has been functioning well. (The lay-out of Reservoir I is shown in Fig. 2.)

RESERVOIR I

Main features of Reservoir I

Area: 18.5 km^2
Volume: 21 million m^3
Average depth: 1.14 m
Mean flow of the feeding River Zala: 9 m^3/s
Retention time: 27 days
Water level of Reservoir I above Lake Balaton mean water level: 2.1 m
Surface area, as percentage of the surface area of Lake Balaton: 3.08%
Shape: triangular or rather fan-shaped
Bottom: mainly silt with varying depth of peat underneath

Main functions of Reservoir I

The main functions are the following:

1. Improving the water quality of River Zala before it is released from the reservoir to Lake Balaton. This corresponds to the function of receiving and treating sewage.
2. Water storage to serve two purposes:
   - regulating the water level of Lake Balaton proper, and
   - flood control.
3. Nature conservation, particularly the protection of birds.
4. Fishing, which, in that case, means well controlled commercial fishing.
5. Reed production for construction purposes and other industrial use.
6. Tourism in the area around the reservoir, and bird-watching.

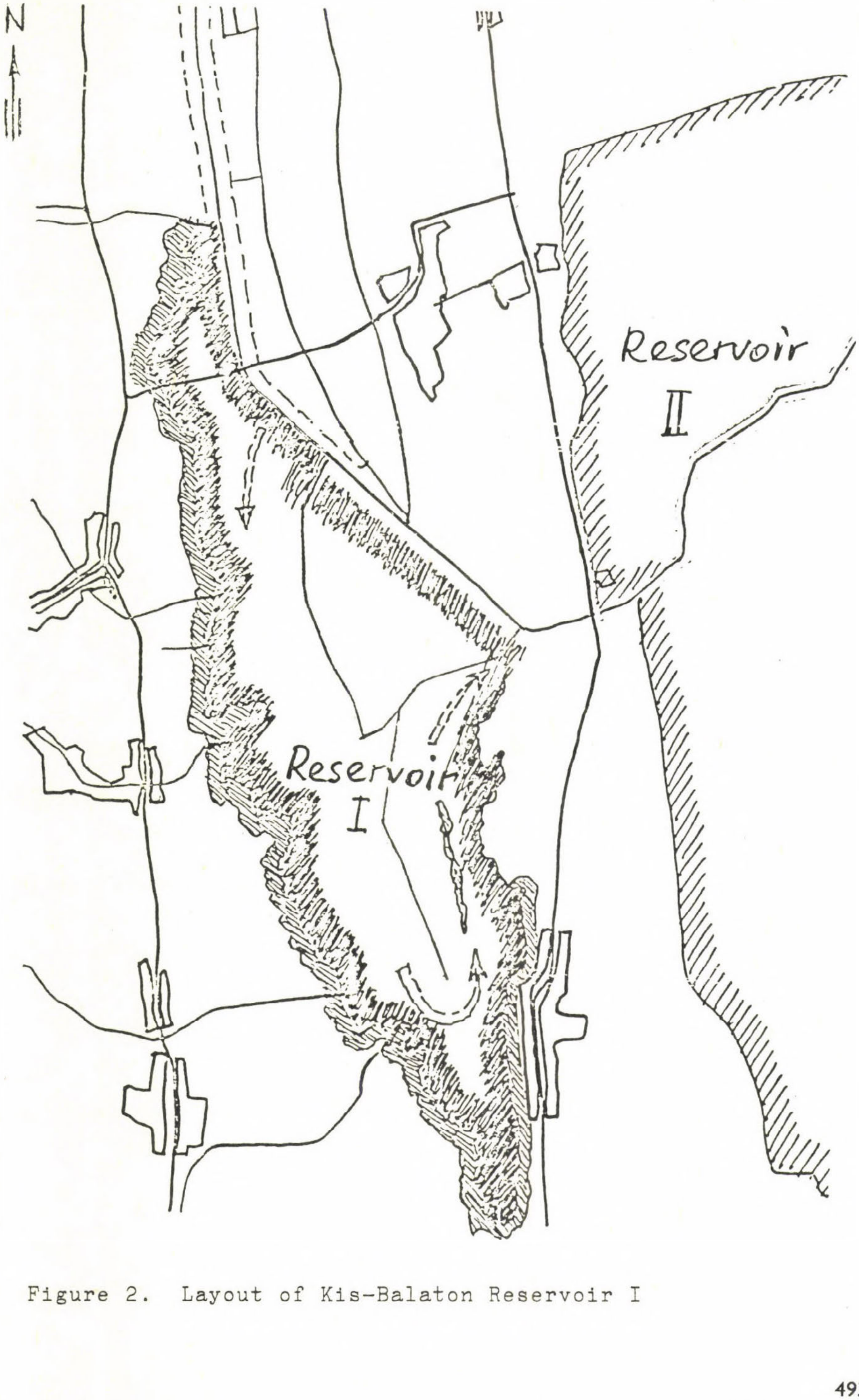


Figure 2. Layout of Kis-Balaton Reservoir I

Interrelationships of the above functions are shown in Fig. 3.

MANAGEMENT REQUIREMENT AND CONSTRAINTS

Both the water purifying and the nature conservational functions of the reservoirs require an undisturbed, quiet environment. To best suit to this requirement, the whole Kis-Balaton, including both Phase I and the yet uncompleted Phase II, have been declared in 1986 a national reserve, the Kis-Balaton Landscape Protection Area.

In accordance with the rules and regulations in respect of landscape protection areas, the following requirements were specified for the whole national reserve:

- No swimming, sailing, boating or other recreational activities are allowed in the reservoirs.
- Angling is prohibited in the reservoirs, except at one particular place, at the northernmost corner of Reservoir I, at a road-crossing.
- Commercial fishing is permitted, but is subject to nature conservation, particularly to the protection of birds.
- Reed cutting or willow cutting can only be done according to the advice of the Landscape Protection Area personnel.
- Regulation of the reservoir's water level is done according to the requirements of Lake Balaton proper. Nevertheless, even this must be done in accordance with the interests of nature conservation at the reservoirs.
- No unauthorized person is allowed to enter the Landscape Protection Area.
- Bird-watching has been made possible at an observation tower, specially constructed for this purpose.

THE CONCEPT OF POST-PROJECT ANALYSIS

Post-project Analysis (PPA) is the last phase of the method

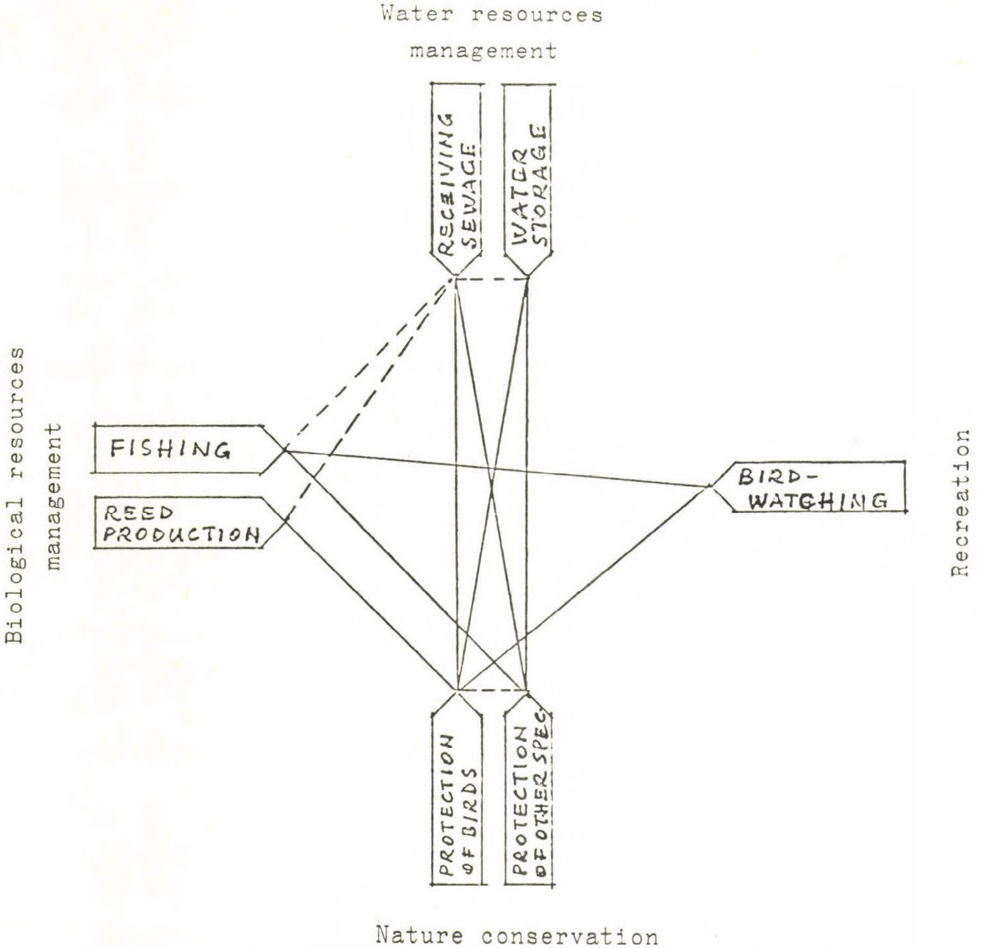


Figure 3. Interrelationships of various functions and uses of Kis-Balaton Reservoir I

Note: Full lines represent conflicting interests and dotted lines non-conflicting interests of Environmental Impact Assessment (EIA), which is normally made up of the following stages:

- Screening to determine whether or not a (full-dressed) EIA is required.
- Scoping to work out the significant alternatives and identify their impacts on the environment.
- Outside review as a quality check of the informations to be presented to the decision-makers.
- Public participation to consult the public at an early stage of the planning process.
- Management of uncertainty to cope better with risks and uncertainties.
- Monitoring of all environmental changes brought about by the construction and subsequent functioning of the project.
- Post-project analysis which, based on collected (monitored) data, makes an assessment of the environmental changes in order to mitigate the harmful environmental effects of the project. PPA is also a useful tool for improving the technique applied to Environmental Impact Assessment.

ENVIRONMENTAL IMPACT ASSESSMENT OF THE KIS-BALATON RESERVOIR

To estimate the environmental impacts of the proposed Kis-Balaton Reservoir, the new planning tool, the above mentioned Environmental Impact Assessment was used.

It was the first time in Hungary that an EIA - even if not a full-dressed one - was worked out for a water development project. (This was followed later on by a much more detailed EIA for the Gabcikovo-Nagymaros Hydro-electric Scheme at the Hungarian-Czechoslovakian border on River Danube.)

Alternatives of the EIA of the Kis-Balaton project

Because of the special case of reconstructing a once existing lake, the question of true alternative proposals - an import-

ant element of the EIA process - did not arise. Nevertheless, there was a possibility of making alternatives depending on time rather than on technical solution or location.
Therefore, apart from the basic variant of constructing the whole project at the same time as one single work, a two-phase alternative was also worked out with a seven year time lag between the two phases.
The latter variant was accepted with completion dates of 1985 and 1992 respectively.

Apart from the financial benefit of smaller yearly amounts of investment, the main advantage of this alternative was the possibility to make improvements in the design of the second reservoir (Reservoir II), based on the experience gained at Reservoir I.

Impact predictions

The Environmental Impact Assessment, which was made to cover both Phase I and Phase II Reservoirs, included the following impact groups:

1. Water quality changes, including physical/chemical and biological characteristics.
2. Water-logging around the reservoirs.
3. Increase of fogginess in the surrounding area.
4. Increase in the number of species and individuals of both animals and plants living in and around the reservoirs, the birds being of particular importance.
5. Nuisance due to the proliferation of mosquitoes in the area.
6. Reduced employment opportunities in the neighbouring villages, due to loss of arable land to be covered by the reservoirs. Although, this problem could partly be compensated for by new job offers in the reed and willow based local industries.

Of these effects nos. 1, 2 and 3 are physical/chemical impacts,

nos. 4 and 5 are ecological (biological) impacts, and no. 6 belongs to the social impacts, according to the impact categorization normally used in the environmental impact assessment practice, that is shown in Fig. 4.
Due to the inaccessibility of the reservoirs for the general public, the aesthetic impacts were not considered.

To illustrate the whole set of environmental impacts, matrices have been used in the EIA, based on the well-known "Leopold matrix", a simplified form of which is shown in Fig. 5.

However, it should be borne in mind that matrices and other usual methods of Environmental Impact Assessment cannot include secondary and tertiary effects. Therefore, in case of complex projects corrections may be required for these indirect effects. A system of the environmental impacts having effects on each other, and finally on the proposed project itself is presented in Fig. 6.

POST-PROJECT ANALYSIS OF RESERVOIR I

Following the completion and start of operation of Reservoir I - the first unit of the Kis-Balaton Project - research on the environmental changes has also begun.

Based on the research findings available up to date, a post-project review could be drawn up on the lines of the EIA of Lake Kis-Balaton, as discussed above.
Main conclusions of this review were the following:

1. There is a definite improvement in the quality of the water in the already existing Reservoir I. However, it is too early to arrive at any conclusion as to the degree of the possible final improvement.

2. Proliferation of animal and plant species was left for nature at Reservoir I. Yet, the colonization of the reservoir by waterfowl was very fast indeed.

   New bird species nesting at Reservoir I in 1987, as reported by our colleague on the site (Futó, E.), were the following:

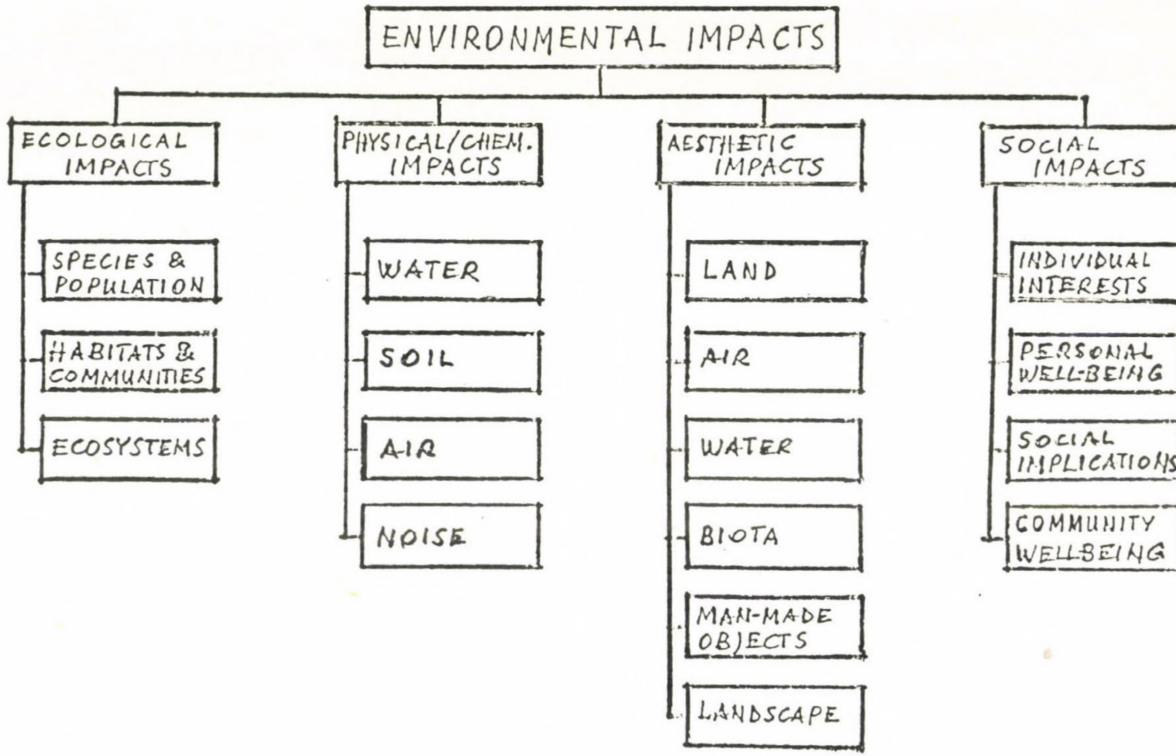


Figure 4. Environmental categories and components

IMPACT VALUES FROM 1 TO 10 / VALUES OF IMPORTANCE OF THE AREA FROM 10 TO 1			PROPOSED ACTIONS													
			MODIFICATION OF THE REGIME OF THE RIVER				TRANSFORMATION OF THE AREA									
			LAND SURFACE	HYDROLOGY	RIVER BED	FLOW	DAM	POWER ST.	SHIP LOCK	UNDERGROUND STR.	ROADS	TRANSMISSION LINES	EMBANKMENTS	PORTS	TOWN DEVELOPM.	RECREATION
ENVIRONMENTAL CHARACTERISTICS	PHYSICAL/CHEMICAL FACTORS	AIR														
		WATER														
		LAND														
		NOISE AND VIBRATION														
	ECOLOGICAL FACTORS	FLORA														
		FAUNA														
	AESTHETIC FACTORS	NOT PROTECTED LANDSCAPE														
		PROTECTED LANDSCAPE														
		MAN-MADE OBJECTS														
	SOCIAL FACTORS	INDIVIDUAL INTERESTS														
		COMMUNITY WELL-BEING														
TOTAL VALUE																

Figure 5. Leopold matrix (for dams)

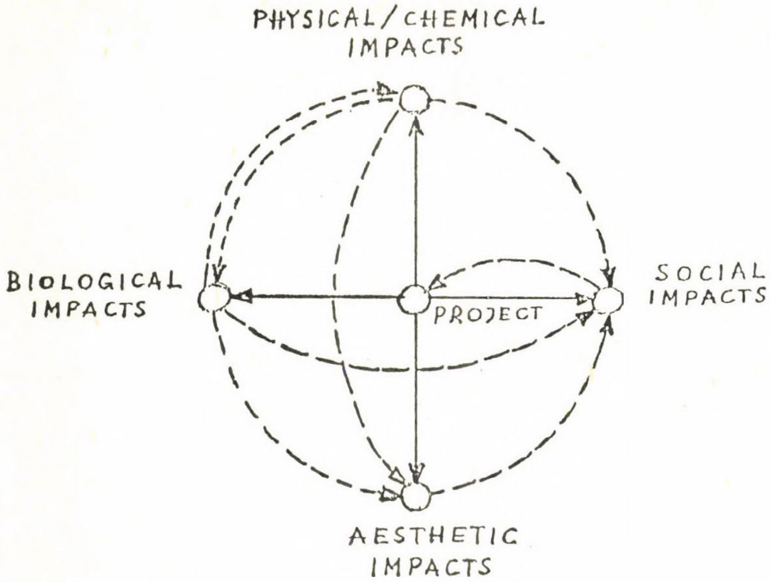


Figure 6. System of environmental impacts of a proposed project

Tufted Duck (Aythya fuligula), Red-crested Pochard (Netta rufina), Mediterranean Gull (Larus melanocephalus), Whiskered Tern (Sterna hirundo).

There was also an increase in the number of individuals of the following existing species:

Little Grebe (Podiceps rufficollis), Black-necked Grebe (Podiceps nigricollis), Great Crested Grebe (Podiceps cristatus), Red-necked Grebe (Podiceps griseigna), Little Egret (Egretta garzetta), Great White Egret (Egretta alba), Spoonbill (Platalea leucorodia), Glossy Ibis (Plegadis falcinellus).

As for the botanical changes, they were also significant, as reported by a specialist of our local office (Pomogyi, P. 1985-86).

By September, 1986, two years after the start of filling the reservoir, 1146.5 ha of its surface area (54.6%) was covered by aquatic plants:

- reed beds occupied 63.5 ha (3% of the reservoir),
- sedge beds occupied 117.8 ha (5.6% of the reservoir),
- 965.2 ha (46% of the reservoir) was colonized by other aquatic macrophytes, mostly by pondweeds, the most numerous of which was Ceratophylletum submersi, covering 437.2 ha, that is 20.8% of the total surface area of the reservoir.

3. Water-logging around Reservoir I has given the Landscape Protection Area personnel some problems, though it did not exceed much the expected level.

4. Fogs around the area have not increased lately. This effect, at least as far as Reservoir I is concerned, does not seem to be a serious problem.

5. Reed and willow based local industries were not developed in the area, still there has been no serious problem with employment as yet.

   However, the Landscape Protection Area personnel is faced with an uncalled for poaching for fish at the shores close to the villages along Reservoir I.
   It appears that some of the old trades including even the illicit ones, can revive in no time once circumstances permitted.

6. Up till now, there have been no complaints by local residents about the increase of mosquitoes in the area. However, a forest belt, planted around Reservoir I to help dry the land around the reservoir may prove to be very useful also in avoiding the occurrence of the mosquito problem later on.

CONCLUSIONS

The Post-project Analysis of Reservoir I of the Kis-Balaton project directs attention to the importance of posterior environmental evaluation of reservoirs in general.

The best way to make these evaluations is to compare the final environmental effects with the expected ones envisaged in an EIA. Hence a full-dressed EIA, corresponding to the latest research findings is the most suitable basis for the PPA of a dam (reservoir) project. However, not having a complete and "perfect" EIA should not discourage one from working out a post-project review. In fact, a Post-project Analysis should be prepared to all existing large-sized reservoirs, independent of the availability or otherwise of a formal EIA in respect of them.

Some sort of environmental impact assessment has certainly been worked out for every dam (reservoir) project anyway, even if those evaluations were not made to follow the present system of the EIA, and were not called by this name.

However, by a careful review of the original designs, a special EIA can be reconstructed posteriorly to all major dams which were designed and constructed without a proper EIA. This special EIA can demonstrate the merits and demerits of the former planning procedure in respect of environmental impact evaluations of such projects, and can also serve as the basis of the Post-project Analysis to be made for them.

This way, instead of heated debates, based on incomplete or biased information, an unprejudiced evaluation can be made of the environmental effects of reservoirs and lake development projects, relying upon the scientific analysis of design figures on one side, and the actual environmental data of the functioning project on the other.

Symp. Biol. Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

MACROPHYTE COMMUNITIES OF THE KIS-BALATON RESERVOIR

P. POMOGYI

Kis-Balaton Department, District Authority Environment
Protection and Water Management
Csik F. sétány 1, H-8360 Keszthely, Hungary

INTRODUCTION

One of the most serious problems in the deterioration of the water quality of Lake Balaton is the eutrophication being especially strong in the Keszthely Bay. The greatest transporter of anorganic nutrients causing the eutrophication is the River Zala flowing into the Bay.

At least half of the nutrients is diffuse in origin, so they cannot be removed before entering the river by the traditional methods (Joó, Lotz 1980).

Therefore, the conception has been formed that the transport of the nutrients should be restricted not far from the mouth of the river by the Kis-Balaton Reservoir. The upper basin has already been functioning since 1985.

One of the biological bases of the functioning of the Reservoir is that in a shallow pond aquatic and wetland macrophytes can multiply and accumulate nutrients excluding them from the biogeochemical cycle for a certain period, and by harvesting they, too, can be removed from the system.

Actually it is the only possibility for nutrient removal. Therefore, special emphasis is focussed on the botanical and plant-ecological investigations in this area.

Studies on macrophyte communities started already in 1982, before the flooding of the Reservoir.

METHODS

Botanical research was carried out on the extent of areas covered by different macrophyte communities, structural changes in the communities, removable biomass and nutrient amounts, the harvested parts of plant-dominated areas and the removed amounts of biomass and nutrients.

The areas of macrophyte communities were measured on vegetation maps, made by using infrared aerial photos. The photos were taken and the vegetation maps, and territorial calculations were made by the Remote Sensor Centre (Budapest), while identification of the plant communities was carried out on the spot by the Kis-Balaton Department (Keszthely).

Between 1985 and 1987 aerial photos were taken and vegetation maps made twice yearly, in early summer and early autumn. It was necessary because of the quick changes in the vegetation structure in the first years after the flooding. From 1988 onward one vegetation map in a year seems to suffice to characterize the slower changes.

Investigations on the biomass and chemical composition of macrophytes are carried out by the traditional methods, published earlier several times (e.g. Kárpáti et al. 1983, Pomogyi et al. 1983, Pomogyi, 1985).

RESULTS AND DISCUSSION

State before the flooding

According to official data calculated by the agricultural field register, an area of 2376 ha was expropriated by the Water Authority as shown in Table 1.

Table 1. Expropriation at the Kis-Balaton Reservoir

Line of cultivation	Per cent	Hectare
Fieldland	8	190
Forest	15	357
Grassland	73	1735
Other (e.g. ditches, etc.)	4	94

Reality somewhat differed from the agricultural field register. Fieldland cultivation could be found only at the higher parts of the area recently being mostly islands. The major part was registered as grassland for grazing and haymaking, however, it was covered by floodland weed stands, and Deschampsia- or Carex-dominated communities, as shown in Table 2.

Table 2. Area of plant communities according to the results of vegetation mapping

Line of cultivation/plant stands	Area (ha)
Cultivated fieldland	140
Uncultivated fieldland	50
Planted forest, forest-belt	373
Salicetum albo-fragilis	3
Calamagrosti-Salicetum cinereae	7
Scirpo-Phragmitetum	25
Agrostietum albae	120
Alopecuretum pratentis	148
Deschampsieetum caespitosae	224
Rubo-Solidaginetum	255
Caricetum	886
Festuca arundinace (planted)	9
Other	136

There was a rather dense network of ditches in the area because of the former efforts to pursue agricultural activities here. The ditches were totally covered by aquatic and marshland macrophytes.

Since, apart from the ditches, the original vegetation was neither a real aquatic nor a helophyte one, formation of marshland communities takes a longer time. Nevertheless, it was necessary to flood this area, too, because of the geography of the Zala basin. If only the lower basin had been flooded, the area of the earlier upper basin would have got under water, too, due to the damming up.

The first changes after the flooding

Flooding of this area started in spring 1984, and took place in five steps up to the early summer of 1985 when the

Reservoir started to operate. In this way water vegetation could form gradually, in two main directions:

1. Where the former plant stands were harvested and removed, the pioneer communities belonged to the Polygonum-dominated ones, such as Polygonetum amphibii ceratophylletosum submersi, and polygonetosum. Polygonum amphibium is an amphibious species, preferring wetland biocoenoses, so after the flooding there were favourable conditions for its predominance.

2. Where the original vegetation could not be removed, and water depth was greater (more than 50 cm), the pioneer community was the Lemno-Utricularietum glycerietosum or phalaridetosum, followed by floating "lawn" of Glyceria maxima, and Phalaris arundinacea. In these stands the biomass reached an incredibly high degree: for example in the Lemna trisulca stand 47 $kg \cdot m^{-2}$ fresh weight and in the floating Glyceria and Phalaris stands 30-40 kg were measured.

At the unharvested shallower parts of the Reservoir the former communities (mostly Caricetum) changed rather fast through a Lemno-Utricularietum phase in the direction of a Typha-dominated vegetation.

After the first two years, based on botanical investigations, 10 "water bodies" could be distinguished where both the trends in the changes of macrophyte community structure and the quantities of accumulated biogenic elements were different. Originally, these water bodies were delimited from each other either by a dyke, by a kind of road or else by a natural (row of hillers) or artificial boundary.

The gradual flooding together with different water qualities caused the differences in the macrophyte stands (Pomogyi 1986).

The first changes in the flora occurred very quickly.

Changes after putting the Reservoir into operation

The Reservoir started to operate in its final form in June 1985, and, as a result, considerable ecological changes occurred. One of the most important changes was that the water started to flow (although at a rather low speed at most places). Another one was that the black colour of the water caused by humic acids (it is a peatland area) started to be diluted by the water of the River Zala.

The drifting ice in winter, alterations in the water level, some construction works carried out in the Reservoir causing changes in flow conditions, together with other inanimate phenomena had markedly affected the macrophyte stands, especially the floating ones.

There were, of course, a lot of changes in the biological, and the nutrient conditions, too, e.g. algal blooms, etc.

The most important changes in vegetation (Table 3) were as follows:

In the early summer of 1985 more than 1000 ha were covered by macrophytes, decreasing slightly on the second observation.

The area of the floating Glyceria- and Phalaris-dominated stands largely decreased, especially in the so-called Casette, where the water of River Zala flows through quickly. The Polygonum-dominated stands started to increase. This increase can be seen nowadays, too.

In the blackwater areas, where the river could only slowly affect Glyceria, Phalaris stands were followed by those of Ceratophyllum submersum, with an enormous biomass during the two years. Since 1987 Polygonetum amphibii has started to develop here.

Between the two mappings in 1985 the area covered by Typha-dominated stands increased, too, while that covered by different subassociations of Scirpo-Phragmitetum somewhat decreased.

By 1986 Phragmites-, Typha-, Polygonum- and Ceratophyllum-dominated stands increased, some subassociations disappeared, Caricetum decreased, and Myriophyllo-Potametum started to develop.

In 1986 more than 1000 ha were covered by macrophytes making up about half of the total area of the upper basin. In early winter that year it could be seen that the proportion of Ceratophylletum submersi decreased, as shown by the data of Table 3. There was a decrease of about 400 ha in the area covered by macrophytes, and it was due to Ceratophylletum.

Table 3. Area of plant communities of the Kis-Balaton Reservoir 1985-1987 (ha)

Associations/subassociations	1985 I	1985 II	1986 I	1986 II	1987 I	1987 II
SCIRPO-PHRAGMITETUM						
phragmitetosum	53.0	58.0	63.0	64.0	64.4	69.0
polygonetosum	0.7	0.3	-	-	-	-
glycerietosum	7.2	1.1	-	-	-	-
sparganietosum	0.5	0.5	-	-	-	-
caricetosum	-	-	-	-	8.7	8.7
Phragmites-dominated total	61.4	59.9	63.0	64.0	73.1	77.7
typhetosum	16.7	15.6	28.1	32.9	35.7	37.6
typheto-caricetosum	3.4	13.5	45.1	46.0	76.5	115.3
Typha-dominated total	20.1	29.1	73.2	78.9	112.2	152.9
CARICETUM ACUTIFORMIS-RIPARIAE						
caricetosum	103.8	83.1	77.6	76.6	46.1	23.9
typhetosum	22.9	18.8	-	-	28.1	28.4
glycerietosum	71.6	65.9	43.0	42.0	66.3	55.6
juncetosum	16.5	11.2	-	-	-	-
phragmitetosum	1.9	1.7	-	-	-	-
Carex-dominated total	213.4	180.7	120.5	117.8	143.8	110.0
LEMNO-UTRICULARIETUM glycerietosum	260.0	153.0	-	-	-	-
GLYCERIETUM MAXIMAE	-	-	32.0	24.0	13.7	6.6
PHALARIDETUM ARUNDINACEAE	31.6	4.0	-	-	-	-
POLYGONETUM AMPHIBII						
polygonetosum	129.0	206.0	139.0	255.0	120.9	196.0
ceratophylletosum	7.7	47.0	120.0	134.0	122.2	208.7
Polygonum-dominated total	136.7	253.0	259.0	389.0	243.1	405.3
CARETOPHYLLETUM SUBMERSI	300.0	300.0	427.0	437.0	7.7	0.7
ANACHARIDETUM CANADENSIS	11.3	11.3	11.3	10.0	-	-
MYRIOPHYLLO-POTAMETUM	-	-	28.0	16.0	3.5	-
OTHER	17.0	17.0	11.2	11.9	19.3	21.3
TOTAL	1060	944	1023	1147	617	775

The areas covered by Phragmites-, Typha- and Polygonum-dominated communities have shown a continuous increase up to now, although the vegetation map from 1988 is not yet completed.

The Carex-dominated stands keep on decreasing currently, too.

In 1987 a new community, <u>Nymphaeetum alboluteae</u> appeared in a five-hectare area. Some ditches had been covered by it before the flooding. With a change in the direction of water flow the Nuphar and Nymphea species could be represented by some specimens. Now they form beautiful stands again.

In 1986 the zonation characterizing the shallow lakes in Hungary started to appear at the edge of the water with communities of Magnocaricion, Agrostion, etc.

In the recent two years the changes in vegetation structure seem to have become much slower.

At the deeper areas of the Reservoir the dominant community is <u>Polygonetum amphibii polygonetosum</u>. Some signs of development have shown that the next community in the future will be <u>Myriophyllo-Potametum</u>, however, this needs a longer period to develop.

At the shallower parts the Typha species will dominate; their area has increased rather rapidly.

The area of Phragmites-dominated stands seems to increase gradually.

These results show that from 1988 onwards, one vegetation map in a year will suffice to follow the changes in the structure of the vegetation at the upper basin of the Kis-Balaton Reservoir.

<u>Biomass, N and P contents of the macrophytes of Kis-Balaton</u>

It is important in the practical management to know the amount of the removable biomass and nutrient content. The theoretically removable parts and the practically removed ones are measured from year to year.

Table 4 shows the specific fresh and dry biomass of the dominant macrophytes.

Table 4. Specific biomass of macrophytes (t/ha)

Plant species	Fresh biomass	Dry biomass
Phragmites australis	18.00	9.36
Glyceria maxima*	189.50	23.29
Glyceria maxima**	72.40	8.09
Typha sp.	14.14	2.43
Carex sp.	12.10	2.05
Phalaris arundinacea	80.30	10.21
Polygonum amphibium	20.40	2.80
Myriophyllum and Potamogeton sp.	16.30	1.45
Ceratophyllum demersum	43.30	3.30
C. submersum	73.70	5.97
Anacharis canadensis	61.40	5.16
Lemna sp.	44.80	3.08
Nuphar and Nymphea sp.	34.00	3.30

*Floating
**Rooted

Floating Glyceria and Phalaris have a very high specific biomass. Among the aquatic macrophytes the biomass of Ceratophyllum submersum is the greatest followed by Anacharis canadensis.

Considering the areal proportions of the macrophytes, the floating Glyceria stands were found to play an important role in the primary production and nutrient accumulation in 1985 (Table 5). Their importance, however, largely decreased later because of the territorial reduction.

Total biomass and nutrient content of Ceratophyllum submersum also showed a decreasing tendency between 1986 and 1987, still it was of the greatest importance among the macrophytes up to the end of 1987.

The data of Table 5 reveal that a large amount of biomass was produced from 1985 to 1987 showing a decreasing tendency. There was a great decline by 1986, and a smaller by 1987. In the first two years the ratio between the total dry and fresh biomass was 1:10, and it was reduced to 1:6 by 1987. This was due to the proportional increase of the area covered by macrophytes of a higher dry material content.

Table 5. Removable biomass, N and P content of macrophytes (t)

Plant species	Dry biomass			Nitrogen			Phosphorus		
	1985	1986	1987	1985	1986	1987	1985	1986	1987
Phragmites australis	570	594	683	4.5	4.7	8.1	0.3	0.4	0.7
Glyceria maxima	5211	673	696	94.7	12.2	8.2	11.5	1.5	2.2
Typha sp.	40	187	707	0.8	3.5	13.9	0.1	0.5	2.0
Carex sp.	461	358	421	7.4	5.7	8.3	1.2	1.0	1.2
Phalaris arundinacea	324	-	-	5.4	-	-	0.7	-	-
Polygonum amphibium	453	1091	574	12.1	29.4	15.9	1.6	3.9	2.7
Ceratophyllum submersum	2072	3407	2014	45.5	74.6	57.5	7.7	12.6	11.1
C. demersum	56	39	-	1.5	1.0	-	0.3	0.2	-
Anacharis canadensis	58	51	-	1.1	1.0	-	0.2	0.2	-
Lemna sp.	52	-	-	1.8	-	-	0.3	-	-
Myriophyllum and Potamogeton	-	22	1	-	0.5	0.03	-	0.1	0.01
Nymphea and Nuphar	-	-	10	-	-	0.2	-	-	0.03
Total	9297	6422	5106	174.5	133.0	118.2	214.0	20.3	19.9

Table 6. Removed biomass and nutrients (t)

Plant stand	Hectares		Fresh biomass		Dry biomass		Nitrogen		Phosphorus	
	1986	1987	1986	1987	1986	1987	1986	1987	1986	1987
Phragmites	40	34	720	612	374	299	2.95	26.0	0.22	3.50
Carex	30	90	363	1809	61	356	0.98	6.5	0.17	0.86
Typha	20	110	410	2255	70	663	1.22	12.1	0.19	1.60
Polygonum	65	128	1326	2611	183	402	4.91	9.1	0.66	1.54
Ceratophyllum	5	100	369	7058	30	644	0.65	16.6	0.11	3.39
Glyceria	5	-	853	-	105	-	1.90	-	0.23	-
Total	165	462	4040	13600	823	2364	12.60	47.8	1.58	7.70

The total amounts of N and P also showed a declining trend from 1985 to 1987 but their ratio was less than that of the biomass.

Recently, about 100 t N, and 20 t P have been accumulated in the removable plant material at the Kis-Balaton Reservoir.

Harvesting at this area started in 1986, covering 165 ha that year (Table 6).

With 4000 t plant material (800 t in dry weight) 13 and 1.6 t N and P, respectively, were taken out of the Reservoir.

In 1987, from 462 ha with nearly 14,000 t of plant material (dry weight) 48 t N and 7.7 t P were removed. Considering that the River Zala transports yearly about 40 t available phosphorus into the Reservoir, in 1987 nearly 20% of it was removed.

In summary, it can be stated that the aquatic vegetation in the upper basin of the Kis-Balaton Reservoir plays an important role in defining water quality and nutrient contents.

SUMMARY

Studies on the macrophyte communities of the Kis-Balaton Reservoir started already in 1982, before the flooding. At that time it was a wetland area mostly covered by different subassociations of *Caricetum acutiformis-ripariae*, and secondly by *Deschampsieetum caespitosae*. Small parts of it were covered by planted forests, and the ditches by aquatic macrophytes.

After flooding the Reservoir (from 1985), macrophyte vegetation was studied by using infrared aerial photos twice a year. In the first years very quick changes in structure characterized the macrophyte communities.

Based on botanical investigations, 10 'water bodies' could be distinguished at the Reservoir, with different trends in both the changes in macrophyte communities and the quantities of accumulated biogenic elements.

Nowadays about half of the Reservoir is covered by macrophytes belonging mostly to the Scirpo-Phragmitetum and the *Polygonetum amphibii* communities.

The macrophytes play an important role in primary production, and nutrient accumulation, too, so e.g. in 1987, 120 t of N and 20 t of P were accumulated in 5100 t of dry theoretically removable plant material.

Macrophyte harvesting provides the only possibility for the removal of nutrients from the Reservoir.

Harvesting started in 1986 at a smaller scale, while in 1987 from 462 hectares, together with 13,600 t fresh plant material, 48 t of N and 7.7 t of P, were removed from the Reservoir.

REFERENCES

Joó, O., Lotz, Gy. (1980): A Zala folyó szerepe a Balaton tó eutrofizálódásában. Vízügyi Közl. 2, 226-256.

Kárpáti, I., Varga, Gy., Kárpáti, V., Pomogyi, P. (1983): Changes in the primary production of the Kis-Balaton water macrophyte coenoses between 1974 and 1983. Proc. Int. Symp. Aquat. Macrophytes. Nijmegen 18-23 September 1983, pp. 111-116.

Pomogyi, P. (1985): Vizi és mocsári makrofitonok szerepe a Kis-Balaton Védőrendszer működésében. MHT X. Ifj. Napok Keszthely, május 22-23, pp. 38-48.

Pomogyi, P. (1986): A Kis-Balaton Védőrendszer I. ütemének területén végzett botanikai vizsgálatok eredményei. MHT VI. Orsz. Vándorgy. Héviz, junius 18-19. I. pp. 436-447.

Pomogyi, P., Kárpáti, V., Kárpáti, I., Józsa, S. (1983): Release of bioelements during decomposition of aquatic macrophytes from Kis-Balaton. Proc. Int. Symp. Aquat. Macrophytes. Nijmegen, 18-23 September 1983, pp. 182-186.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

HYDROBIOLOGICAL SURVEY OF THE KIS-BALATON RESERVOIR

B. CSÁNYI, P. GULYÁS, J. NÉMETH

Research Centre for Water Resources Development (VITUKI)
H-1095 Budapest, Kvassay Jenő út 1, Hungary

ABSTRACT

Faunistical and algological data were obtained for 11 sites on the Kis-Balaton reservoir by qualitative and quantitative sampling methods since 1985, in spring, summer and autumn.

The heterogeneity of the artificial lake was studied based on the community structure of different compartments (phytoplankton, zooplankton, macro-invertebrates and fishes). The results compared with the conclusions of a chemical-environmental survey at the same sites indicate definite changes in water quality.

The paper summarises the direction of succession and important colonisation phenomena especially at the beginning of the reservoir operation. Multivariate methods (cluster and correspondence analysis) were used to indicate the spatial heterogeneity of different communities.

INTRODUCTION

The Kis-Balaton reservoir system was designed on the main effluent river of the Lake Balaton. The River Zala carries most of the plant nutrient load causing serious eutrophication problems in the last two decades.

According to the original planning, a two-step reservoir system has to be developed on the River Zala, where the P- and N-containing compounds can be removed mainly by sedimentation, adsorption and nutrient uptake of living organisms. The upper reservoir established approximately 10 kms from the Lake Balaton started to operate in 1985 when it was completely filled up; the lower one is still under construction. The main parameters of the system calculated for the average operational water level are summarised in Table 1.

Table 1. Main parameters of the reservoir

	Upper Reservoir	Lower Reservoir	Total System
Surface area (km^2)	18.5	50.9	69.4
Volume ($10^6 m^3$)	21	63.7	84.7
Average depth (m)	1.14	1.25	1.2
Calculated retention time (days)	43	88	132

Average flow of the River Zala (Q)=5.6 m^3/s at Zalaapáti.

A detailed chemical survey of the mass-balance processes and the removal efficiency of the phosphorus- and nitrogen-compounds started in 1985. The project together with hydrobiological investigations started at the same time had two major objecives:

(1) Description of important chemical and biological changes that are immediately connected to water quality in the reservoir;

(2) To determine directives for optimal management of the reservoir on the basis of the results.

This paper deals with the hydrobiological investigations carried out in the reservoir between 1985-88. Only the most important results are presented.

The objectives imply that there are well-defined relations between the water quality status and the species composition of communities. Although the concept of indicator organisms has a long history in pollution biology (Cairns and Dickson 1973; Hart and Fuller 1974) the application of the ecological indicator-indicandum phenomena has many practical and theoretical difficulties. A detailed study on the typification and classificiation of rivers was summarised in the U.K. (Furse et al. 1984; Wright et al. 1984). The conclusions illustrate that the macro-invertebrate fauna has many advantages for indicational purposes. However, the Hungarian water quality management practice completely neglects the observation of this group of species, mostly because there are many taxonomical problems and, at the same time, only a few experts deal with macroinvertebrates.

Depending on the contracts received from the authorities, different emphasis was placed on different communities during the study period. The interpretation of results is difficult because the various groups of species have distinctly different attributes (size, biomass, abundance, etc.). Therefore, basic scale problems arise when comparing these communities. Separated data processing has to be used to avoid misleading conclusions.

METHODS

Plankton samples were collected by the experts of the Western Transdanubian Water Authority (NYUVIZIG) at each of the 11 sites from the surface layer of the reservoir with two weeks and one month frequency between May-October and October-May, respectively. The algal samples were conserved with Lugol-solution and formaldehyde. The volume of water filtered through a plankton net (70 um mesh size) for taking the zooplankton samples was 20 l and 4% formaldehyde solution was applied for conservation.

A pond net (1 mm mesh size) on a handle was used with the kick and sweep technique (Furse et al. 1981) for collecting qualitative macro-invertebrate samples from habitats with abundant macrophytes. An EKMAN-grab was also applied for sampling the soft muddy bottom layer without macrovegetation. The macroinvertebrate sampling points are enumerated in Figure 5. (The numbers are different from the plankton sampling points).

The planktonic species were identified with an Utermöhl-microscope, the invertebrate samples conserved with 4% formaldehyde were investigated under stereomicroscope.

Two 20 m long trawling nets (2 mm mesh size) for the juvenile, three gill nets (10 m, 20 m, 50 m) and occasionally some trapping nets for the adult specimens were employed during the experimental fishing. Presence-absence and relative frequency of species were recorded from different parts of the reservoir. The localization of the sampling sites and the experimental fishing places are shown in Figure 1.

A classification program (NCLAS2) and an ordination program (PRINCOMP) from the SYN-TAX III. package (Podani 1988) were employed for evaluating spatial heterogeneity. Clustering the sample sites with the average linkage method and several distance or resemblance functions was followed by the ordination of the same data set using correspondence analysis.

RESULTS AND DISCUSSION

Phytoplankton and zooplankton

The yearly change of the average chlorophyll-a content directly proportional to the algal biomass is illustrated in Figure 2. Rapid planktonic eutrophication can be observed in the whole reservoir, especially in the last two years. The reason of this phenomenon is the high rate of P-load (12-20 mg total $P/m^2/day$). The algal blooms caused by blue-green algae

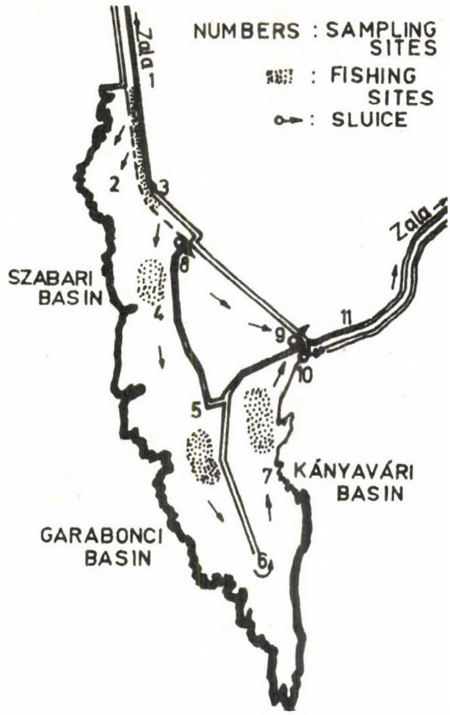


Fig. 1. Kis-Balaton upper reservoir

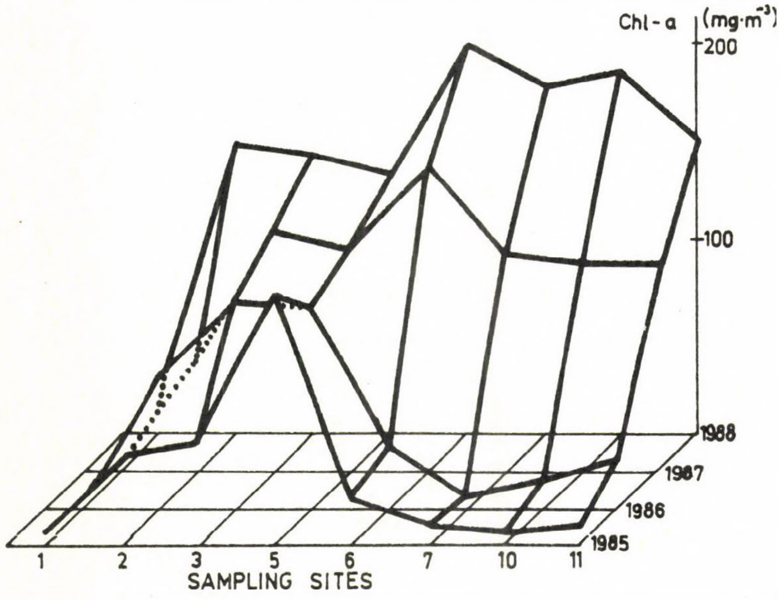


Fig. 2. Yearly average chlorophyll-a content

(mainly Aphanizomenon flos-aquae, Anabaena spiroides, Microcystis flos-aquae) occurred in the middle basin of the reservoir during of the first two years. These phenomena became more frequent in time and space, as well.

Results of qualitative and quantitative investigations were evaluated with cluster analysis for delimiting the different characteristic areas. The southern basin was well separated during the first year. However, since 1986 the large differences in the coexistential pattern of phytoplankton disappeared. Algal blooms caused by blue green algae spread over the whole reservoir paralelly with the decreasing amount of submerged macrophytes (Ceratophyllum demersum etc.).

The change of the zooplankton community structure indicate the same phenomena. The dendrogram of the cummulative quantitative data list in 1987 shows that all of the sampling sites in the reservoir are well-separated from the River Zala (Figure 3).

Since 1986 the Rotatoria population became the most abundant group between the zooplankton. The population size reached the

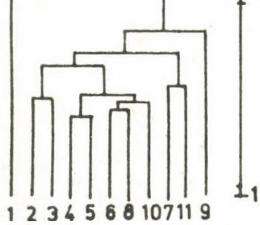


Fig. 3. Dendrogram of cumulative zooplankton data ($D = 1 - S_{\text{Sörensen}}$)

2000 i/l value during the Microcystis aeruginosa bloom in late summer of 1988 and usually the same order of magnitude characterises the community during the last three years.

Macro-invertebrates

The number of macro-invertebrate species found at different basins in 1986 and 1987 is illustrated in Table 2. Presence means that a given species occurred at least once at a given site in 1986-87.

Table 2

	River Zala	Szabari basin	Garabonci basin	Kányavári basin	Total
1986	19	11	25	32	39
1987	40	54	56	69	85

Cluster analysis based on the complement of the Sörensen index was applied to the classification of the sampling sites (Figure 4).

The habitats with abundant macrovegetation (sites 2, 4, 6, 8) are in the same cluster. Another cluster was formed by sites that have pure sediment only (3, 5, 7). The upper section of the River Zala (site 1) appears as an outlier on the dendrogram because of its characteristic bottom-dwelling fauna.

The same data set was subjected to correspondence analysis (PRINCOMP program). On the ordination scattergram (Fig.5) the above three clumps of sampling sites are clearly separated. According to the site-groups there are distinct clusters of species found at different places, too. Cluster A contains 7 species characterising the fauna of the river-sediment. Cluster B incorporates as many as 34 species found in the habitats with macrovegetation. Cluster C has 14 sediment dwelling species from the reservoir.

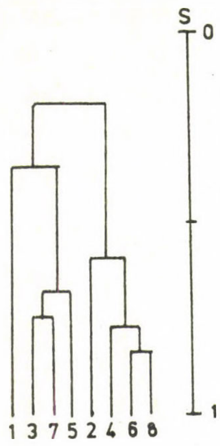


Fig. 4. Dendrogram of macro-invertebrate sampling places ($D = 1 - S_{Sörensen}$)

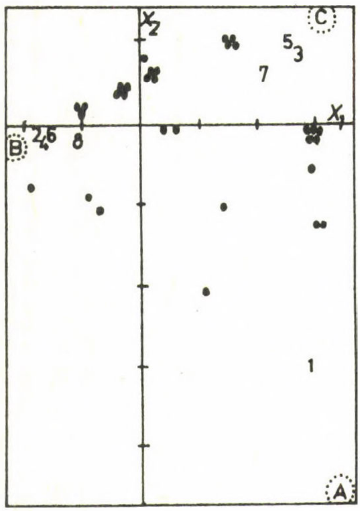


Fig. 5. Ordination scattergram of sampling sites (1-8) and taxa (A-C ●).

Sediment	Macrovegetation
1: Zala	2: Zala
3: Szabari basin	4: Szabari basin
5: Garabonci basin	6: Garabonci basin
7: Kányavári basin	8: Kányavári basin

After the ordination the transposed data matrix was analysed by the NCLAS2 clustering program. In this procedure the species were the objects and the sites were the attributes. On this basis another dendrogram was constructed (Figure 6).

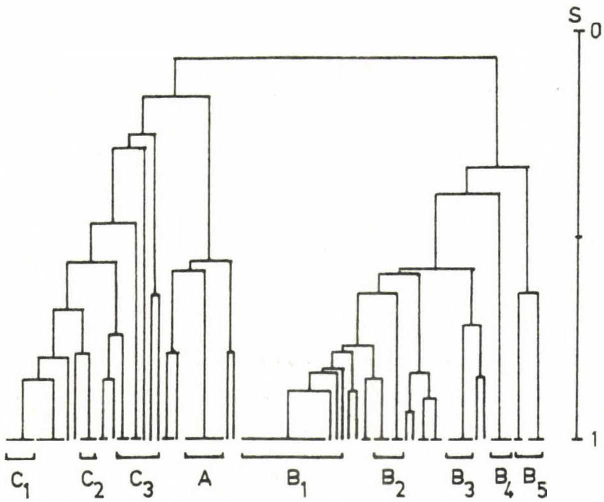


Fig. 6. Dendrogram of macro-invertebrate taxa
($D = 1 - S_{\text{Sörensen}}$)

According to te comparison of ordination and classification methods the following species groups can be recognised:

<u>GROUP A</u> (Species from the River Zala)

Anodonta cygnea, Pisidium amnicum, Pisidium henslowanum, Sphaerium rivicola, Theodoxus danubialis, Astacus sp., Gammarus roeseli.

<u>GROUP B</u> (Species from macrovegetation)

Group B_1 (Over the whole reservoir)

Erpobdella octoculata, Helobdella stagnalis, Plumatella fungosa, Lymnaea peregra, Lymnaea stagnalis, Planorbarius corneus, Agrion (Coenagrion) sp., Libellula sp., Platicnemis pennipes, Hydrometra stagnorum, Gerris sp., Micronecta sp., Notonecta glauca, Sigara sp., Haliplus sp., Ilybius sp.

Group B_2 (Except the River Zala)

Planorbis carinatus, Naucoris cimicoides, Nepa cinerea, Theromyzon tessulatum, Dytiscus marginalis.

Group B_3 (From Garabonci and Kányavári basin)

Aeshna sp., Erythromma najas, Plea leachi, Ranatra linearis, Laccophilus sp.

Group B_4 (From Kányavári basin only)

Glossiphonia heteroclita, Gyrinus natator, Psectrocladius sp., Rheotanytarsus sp.

Group B_5 (From the middle region, the Szabari and Garabonci basin)

Hemiclepsis marginata, Viviparus contectus, Hirudo medicinalis, Succinea elegans, Hydaticus sp.

GROUP C (Species from the sediment)

Group C_1 (Over the whole reservoir, except the river)

Ophidonais serpentina, Stylaria lacustris, Acroloxus lacustris, Succinea oblonga, Tanypus punctipennis.

Group C_2 (From Szabari and Kányavári basin)

Ceratopogonida sp., Chaoborus flavicans, Tanypus villipennis.

Group C_3 (Mostly from the Kányavári basin)

Hippeutis complanatus, Limnophilus flavicornis, Armiger crista f. nautileus, Armiger crista f. cristatus, Valvata cristata, Lymnaea truncatula.

Fishes

The fish fauna was investigated only in the first two years of operation at four different sites (Fig.1). The subsequent experimental fishing provided interesting relative frequency data of species in the first year, when a sudden expansion of the crucian carp population was observed over the whole reservoir. The sequence of abundances in the total catch expressed by relative frequency is the following: Carassius auratus gibelio (43.9%), Cyprinus carpio (20.9%), Tinca tinca (8.6%), Esox lucius (2.6%).

Whereas 13 species detected in 1985, 19 species were found next year. Both the carnivorous and cyprinid species increased in number. The upper Zala-section and the final basin (Kányavár) had 13 fish species, the Garabonci basin with hypertrophic conditions contained only 9 species. The most abundant fauna was found in the Szabari basin having most of the open water surfaces (17 species). Serious water quality problems were indicated in the exceptionally dry and hot summer of 1988. Fish- and water bird species died in several basins (algal bloom and Clostridium bacteria were registered). Large amounts of algal biomass left the reservoir through the outlet sluices and loaded the lower river section with organic and nutrient material.

The results of the macro-invertebrate study indicate that the areal distribution of species is determined by the habitat types, i.e. similar species groups can be found in the same habitants of different basins.

REFERENCES

Furse, M.T., Moss, D., Wright, J.F., Armitage, P.D. (1984) The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. Freshwater Biology 14, 257-280.

Furse, M.T., Wright, J.F., Armitage, P.D., Moss, D. (1981) An appraisal of pond-net samples for biological monitoring of lotic macroinvertebrates. Water Research 15, 679-689.

Cairns, J., Dickson, K.L. (1973) Biological methods for the assessment of water quality. ASTM Special Technical Publication, 528.

Hart, C.W., Fuller, L.H. (1974) Pollution ecology of freshwater Invertebrates. Academic Press, New York etc.

Podani, J. (1988) SYN-TAX III. Users manual. Abstracta Botanica Vol. 12, Suppl. 1. pp. 1-183.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

RESTORATION OF LAKE VELENCE, HUNGARY

F. SZILÁGYI, Sz. SZABÓ,* M. MÁNDOKI

Research Centre for Water Resources Development (VITUKI)
H-1453 Budapest, Pf. 27, Hungary
*Mid-Transdanubian Water Authority (KDT VIZIG)
H-8001 Székesfehérvár, Pf. 81, Hungary

Abstract

Lake Velence is, with its 25 km^2 surface area, the second largest lake in Hungary. Its main water use is recreation. In the sixties the ever accelerating social-economic development of the catchment basin of 600 km^2 area resulted in the simultaneous acceleration of the upsilting and aging processes of the lake, thus decreasing the lake's recreational value. In order to improve the conditions of recreation the Mid-Transdanubian Water Authority has launched an intensive development programme in the early sixties, involving the dredging of sediments and reed over the larger part of the lake as well as the construction of shore protection works and new ports. By 1987 most of the works have been completed. As the result of technical control strategies the surface area of the lake has been decreased by 1.0 km^2 along with the decreasing of the reeded fraction of the lake surface from 59 % to 43 %. About nine million m^3 soft bed material has been removed as well. Simultaneously with the technical control of the lake research into water quality has been also started. The study gives a summary of the types and technologies of control strategies along with the timing and scheduling of the works and evaluates their effects on the water quality of the lake.

1. Introduction

Lake Velence is the second largest lake in Hungary, both in respect to its surface area (appr. 25 km^2) and to its recreational importance. The lake was formed approximately 10 thousand years ago (Bendeffi, 1969) at the southern foot of the Mid-Transdanubian Mountain Ranges, where it joins the lowland area of Mezőföld. Due to its favourable geographical conditions, and to the closeness of the capital Budapest, this relatively small lake became one of the most popular recreational resorts of the country.

Increased loads of plant nutrients due to the intensive social-economic development of the region accelerated the ageing process of the lake turning it into a swamp-like water body with deteriorating water quality. Recognizing these unfavourable processes the Lake Velence Executive Committee has been established in the late fifties with the objectives of launching a wide-spreading development programme aimed at the halting of the ageing process and reclaiming and utilizing its recreational value. Dredging and shoreline protection works of the past two decades served the interests of improving the water quality in that part of the lake which is devoted to water sports, recreation and tourism. This means that the primary purpose of these activities was not associated with nature conservation. Recreational water uses and nature conservation activities were spatially separated from each other within the lake. A smaller (4.2 km^2) part at the southwestern end of the lake was declared a nature preserve, where no engineering control action has been allowed. Dredging and shoreline training projects were restricted to the eastern part of the lake. This paper describes the lake management actions carried out in the recreational part of the lake focussing on the improvement of water quality.

2. The lake and its catchment basin

At 102.613 meters above Baltic sea level the area of the lake surface is 25.3 km^2. Its length is 10.8 km with 2.3 km average width and 1.45 m average depth. The volume of the lake is $36 \cdot 10^6 m^3$ and the area of the catchment basin is 602 km^2. The ratio of catchment basin area to the water surface is appr. 24. Before lake management actions about 59 % of the lake surface (15 km^2) was covered by reeds. These closed Phragmites stands separated the lake surface into a series of smaller open areas, so-called "clearings", that hardly communicated with each other and had, consequently, much differing water quality. (Felföldy, 1972, 1974). Although the lake is very shallow the reeds prevented wind action from resulting in waves that could cause significant resuspension or currents (Akantisz, 1971).

The geological formations of the catchment basin (Fig.1) are widely varying consisting of fractured palaeozoic granite, trias dolomite and pleistocene loess. An important feature of the basin is that the largest part of the rainfall will not run off into the lake, but infiltrates into the ground, entering also confined aquifers. The special hydrological regime of Lake Velence is due to the above conditions, somewhat in contradiction with the large catchment basin area to lake surface area ratio (24). A fertile topsoil covers most of the basin area, favouring agricultural production and creating a potential for high agricultural land runoff of plant nutrients (Baranyi, 1976).

3. Water household

The multiannual average precipitation onto the lake surface is 613 mm/yr. Due to the presence of large macrophyte masses evaporation is about 10 % higher than that of the open surfaces. Considering the relatively small water volume of the lake this fact is of significance in respect to the water balance. The lake is drained by a single outflow canal, the Dinnyés-Kajtori canal, at an average rate of 414 mm/yr. To maintain constant water level outflow is regulated intermittently thus resulting in lengthened residence time (Baranyi, 1976; Antalné, 1987).

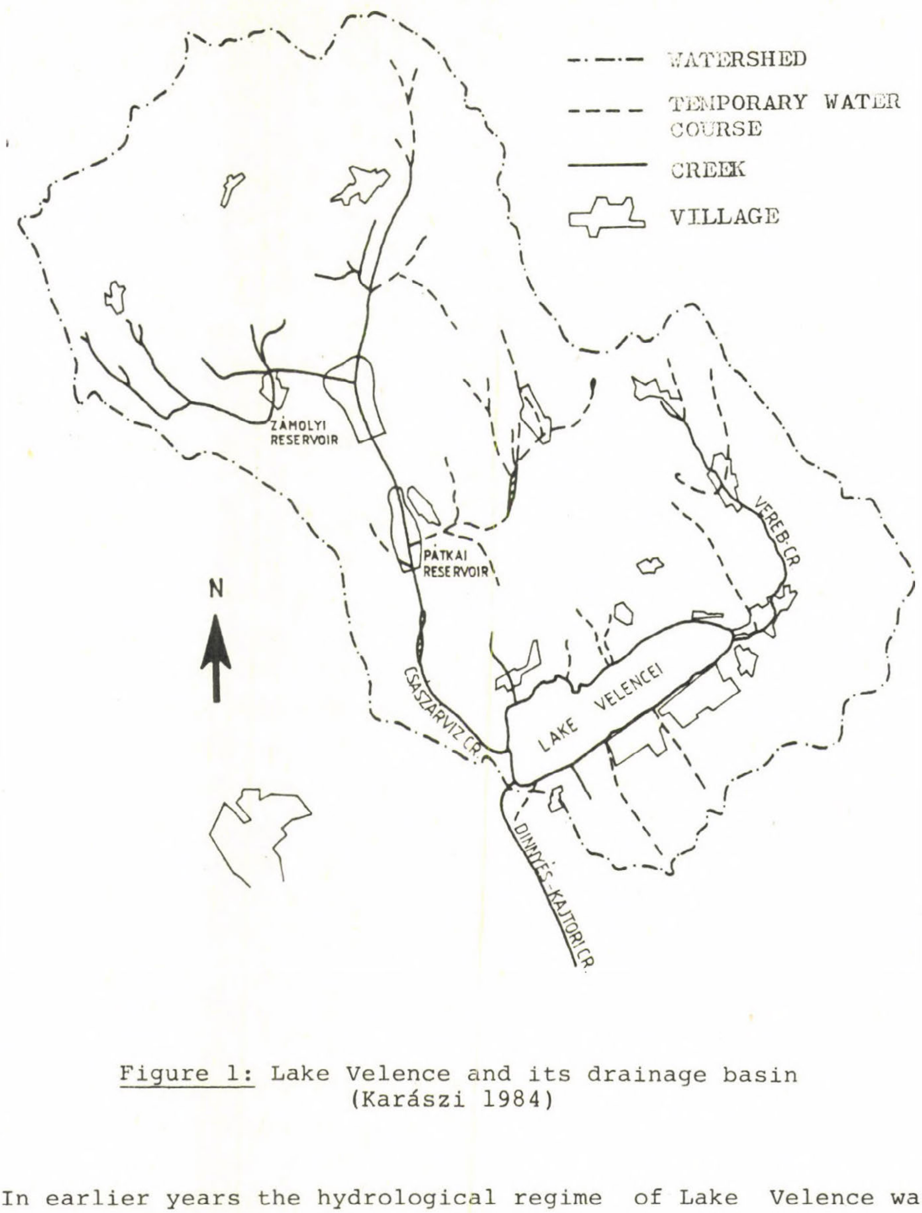


Figure 1: Lake Velence and its drainage basin (Karászi 1984)

In earlier years the hydrological regime of Lake Velence was characterized by extreme conditions. To stabilize water levels two reservoirs (reservoirs Pátkai and Zámolyi) were constructed on one of the major tributary creeks (Császárviz).

Two-thirds of the total catchment basin is represented by that of Császárviz, the largest tributary creek (Q: 0.085-1.7 m^3/s)

entering the lake at its southwestern end through the sudd of the nature preserve, thus relieved from most of its contaminants. The Creek Vereb-Pázmándi at the southeastern end of the lake drains about 20 % of the total catchment, and in terms of flow it is the second largest tributary stream (Tóth,1980,1987).

4. Plant nutrient loads and water quality

On the plant nutrient loads of Lake Velence only scattered data are available. No estimation of the load originating from the lake's direct catchment, caused by depositions from the atmosphere, and represented by unrecorded high flows has been ever made, contrary to the case of Lake Balaton where such estimates were enabled by research (Jolánkai and Somlyódy, 1981). Consequently the loading data of Table 1 are only of informative value, characterizing mostly only the ratio of loads originating from differing sub-basins.

Table 1: Nutrient loading of Lake Velence (mg m^{-2} day^{-1}, 1986)

	NO_3-N	TN	PO_4-P	TP
Vereb-Pázmándi Creek	2.94	3.14	0.016	0.03
Császárviz Creek	1.88	2.37	0.080	0.14
Other creeks	1.01	1.12	0.004	0.01
Total	5.83	6.63	0.100	0.18

About half of the total nitrogen load to Lake Velence originates from the catchment basin of Creek Vereb-Pázmándi, while about one-third from that of the Császárviz. Of the total nitrogen 88% is NO_3-N. One-third of the nitrate load arrives via Császárviz, while half of it is contributed by Creek Vereb-Pázmándi (Tóth, 1981, 1983, 1987).

In respect to phosphorus load Creek Császárviz is the most important contributor, representing about 80 % of the total PO_4-P load. Creek Vereb-Pázmándi yields only about 17% of both the total P and the PO_4-P load. In respect to phosphorus load the other tributaries are negligible. As indicated by the data the nitrogen load to Lake Velence is 37 times higher than the phosphorus load. This difference is even higher if one considers the ratio of biologically available components; the ratio of inorganic-N to PO_4-P is 68:1. This high N/P ratio seems to explain the fact that in the hypertrophic eastern parts of the lake blue-greens did not frequently cause algal blooms.

Before the implementation of lake management strategies the reed cover higher than 50 % resulted in much differing water body patches in respect to both species composition of communities and the chemical water quality. The water of these resource patches was practically not exchanged, due to the much restricted character of currents (Felföldi, 1974). In the internal open patches ("clearings") of the lake inflow had but minor effects and the water quality conditions were mostly formed by the effects of precipitation and evaporation (Baranyi, 1976). By the late eighties intensive dredging has much changed this picture.

Due to the small runoff the conductivity and ion concentration of the lake water are higher than those of the inflowing streams. Conductivity is in the range of 1990-2410 µS/cm, while ion concentrations vary between 1850 g/m^3 and 2270 g/m^3. Major cations are Na^+ (330-340 g/m^3) and Mg^{2+} (150-160 g/m^3), while the major anions are HCO_3^- (680-850 g/m^3) and SO_4^{2-} (380-460 g/m^3). The chemical oxygen demand of the water is relatively high (about 20 g/m^3). Towards the eastern part of the lake the chlorophyl-a concentration of the water is increasing markedly. NH_4-N and NO_2-N concentrations are usually beyond detectability, and NO_3-N is also low, as compared to that of inflowing streams, even in the hypertrophic "clearing" of Fürdető (1.58 g/m^3). Most of the total nitrogen (70-96 %) is organically bound. Orthophosphate phosphorus amounts to 15-55 per cent of total phosphorus.

One of the major causes of increasing nutrient loads was the accelerated social-economic development of the catchment basin area during the past two decades.

5. Socio-economic development in the catchment basin

Although no actual values of the nutrient loads can be estimated on the basis of these data, they certainly do indicate changes of the loading conditions, since development indices are closely related to the nutrient loadings of the lake. Attempts were made to estimate the past loading rates of the lake on the basis of drainage basin development and urban development figures (Statistical Year Books, 1966-1986, Fig. 2).

During the period investigated the permanent population has not changed. Visitor days, however, were increased by a factor of eight and fertilizer usage was increased fivefold as compared to those of the reference year of 1966. Live stock increase rate was only 1.5. In respect to communal facilities the total length of drinking water pipelines was increased fourfold, while that of the sewers was approximately trebled, during the past two decades. Plant nutrient load increases due to tourism and agriculture were somewhat attenuated by the intensive development of sewerage and the diversion of sewage out of the drainage basin.

In Lake Velence systematic hydrobiological research has started in the late sixties in order to provide scientific basis for

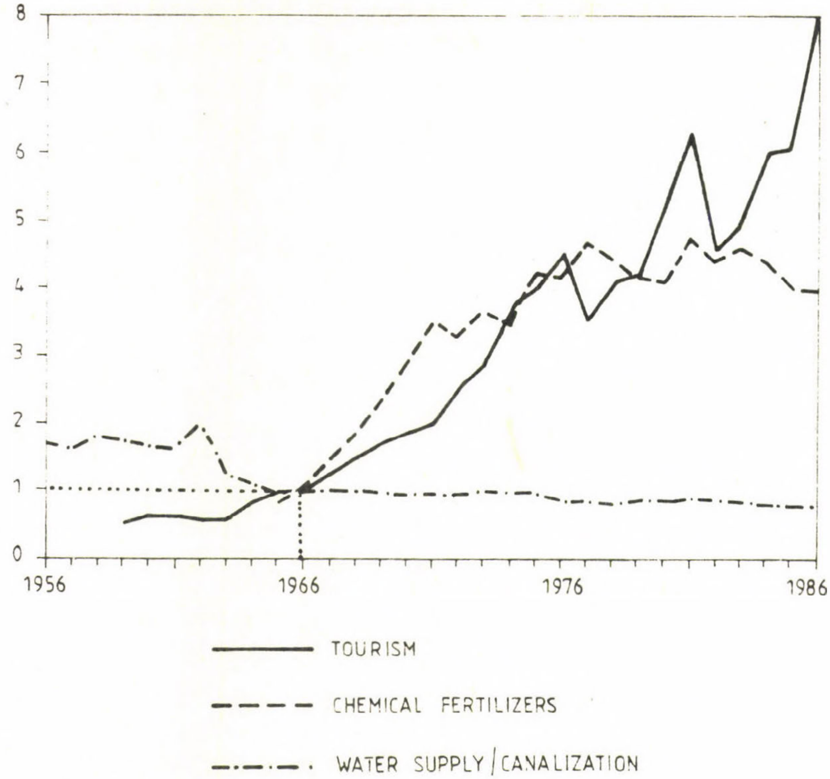


Figure 2: Social-economic development on the drainage basin of Lake Velence

engineering control measures and to follow the hydrobiological and water quality changes of the lake (Felföldy, 1971, 1974). Regular investigations for water chemistry, phytoplankton and zooplankton (Felföldy, 1971, 1972, 1974, 1975, 1977, 1979; Tóth, 1981, 1983) were carried out, along with occasional studies on macrophyta (Felföldy, 1969, 1971, 1972, 1975, 1977, 1979 ; Tóth, 1980; Kovács et al., 1985), bacterioplankton and zoobenthos (Felföldy, 1969, 1971). Sessile communities (Tóth, 1985), fish stock (Felföldy, 1969; Tóth, 1973) and the chemistry of bottom sediment (Felföldy, 1971, Szilágyi, 1982) were also subjected to studies. The investigations were made mostly by the researchers of VITUKI and the Mid-Transdanubian Water Authority (KDT-VIZIG).

Based on these investigations the lake's surface area can be divided into six areas of differing water quality characteristics (Fig.3). The size and ratio of these differing areas are as follows:

I. The nature preserve; dark waters of high humic material content belong to this category. In addition to the area of the nature preserve two other nearby clearings belong to this group (area = 10.9 km^2, ratio 44 %).

II. The Hosszútisztás (long-clearing) of Agárd; the site of former transitional waters (2.4 km^2, 10 %).

III. Nagyviz (the "great waters"), characterized earlier by greyish colour (6 km^2, 24 %).

IV. The boat-racing course, that belonged earlier to the group of greyish waters (1.9 km^2, 8 %).

V. The vicinity of Kárászos, the earlier area of brownish waters (3.0 km^2, 12 %).

VI. The clearing of Fürdető, the area of earlier hypertrophic green waters (0.8 km^2, 3 %).

6. Engineering control strategies implemented

The implementation of engineering control strategies has been started in the mid-sixties. The shoreline protection works and lake bed regulation activities included: bottom silt dredging, reed removal, weed control, shore protection, port construction and local bottom sediment dredging. From the viewpoint of water quality the dredging activities proved to be the most important ones. Dredging operations were carried out in areas shown in Fig.4. The objective of these operations was to improve water quality by removing loose bottom sludge, rich in organic substances, and to maintain an appropriate draft for

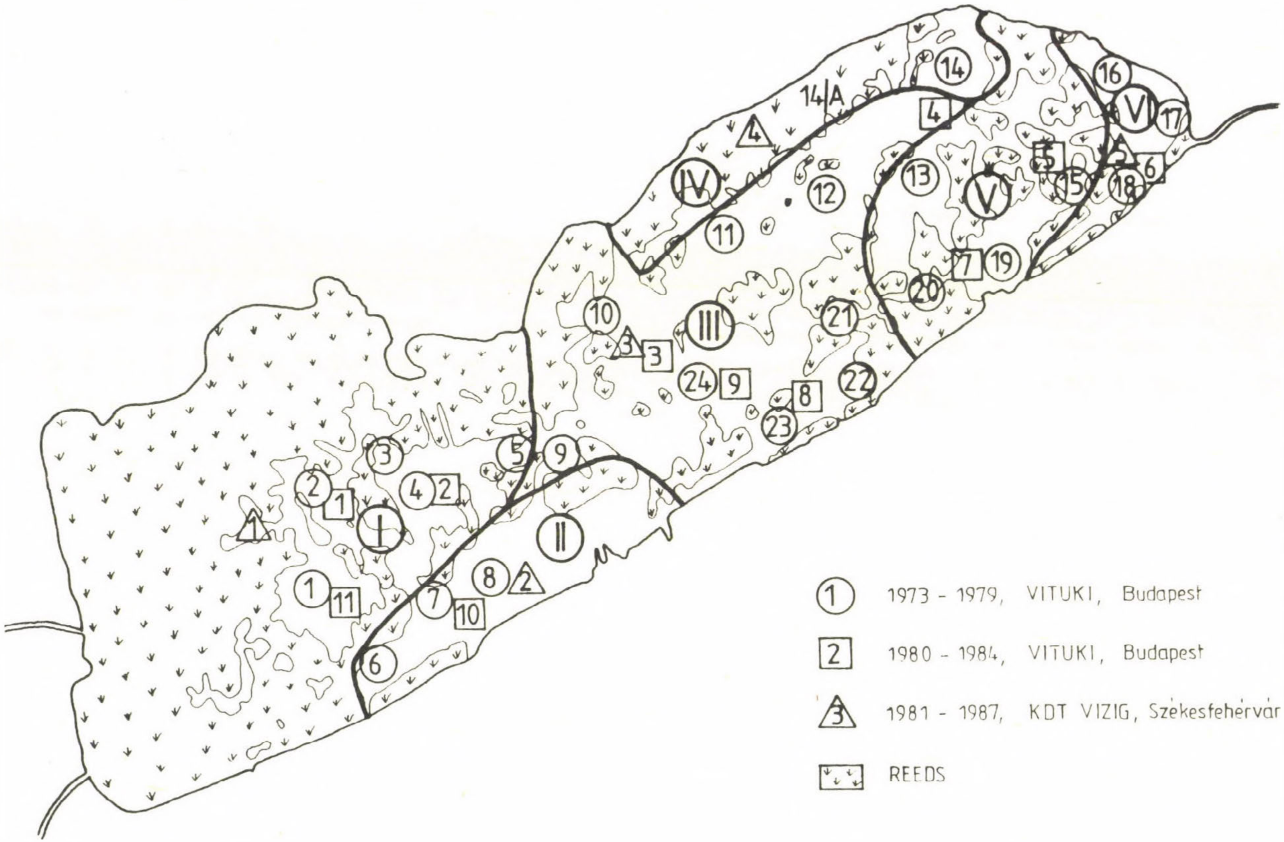


Figure 3: Water quality sampling points and the six lake regions of differing water quality

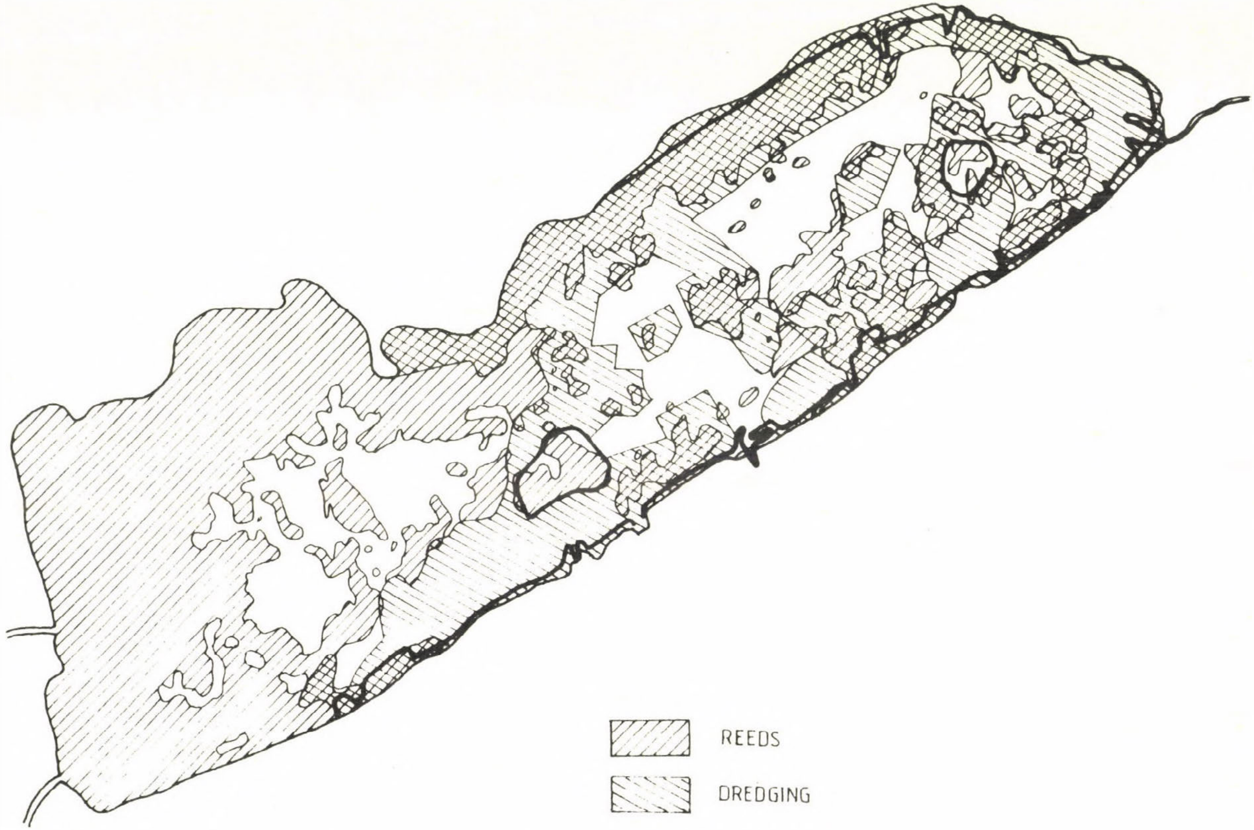


Figure 4: Engineering control measures implemented in Lake Velence

recreational and sport vessels (boating, rowing and sailing). Reed removal by dredging resulted in larger water surface, that in turn enhanced wind induced wave action thus improving the reaeration of water (i.e. oxygen uptake). The main purpose of shoreline regulation was to provide an uninterrupted public shoreline zone along with the associated harbour constructions to facilitate navigation, rowing, sailing and angling.

In course of these operations about 9 million m^3 bottom material, rich in organic substances, has been removed and almost 24 km length of shore protection structures with appropriate boat harbours were built. Of these works 80 per cent were constructed between 1972 and 1982. As a result of these control operations the lake surface area was decreased by 1.0 km^2, while the open water surface of the lake has been increased by 4.0 km^2 (Table 2).

Table 2: Changes in the geometric properties of Lake Velence as affected in dredging

	1959		1987	
	(km^2)	%	(km^2)	%
Area	25.3		24.4	
Open water	10.3	41	13.9	57
Reeds	15.0	59	10.5	43

7. Water quality effects of engineering control strategies

The water quality changes of the lake were followed in terms of the following water quality constituents: chlorophyl-a, Secchi depth, COD_{Mn}, conductivity, total ion content and plant nutrient forms. The regular sampling system of VITUKI and KDT

VIZIG is shown in Fig.3. The number of annual samples varied between 5 and 13.

In order to illustrate the effects of dredging operations two characteristic regions were selected as examples: The first is the formerly fully closed clearing of Fürdető that was a characteristically hypertrophic area (area VI) and the second example is the nature preserve where no dredging operations have been carried out (area I, see Fig.3).

The changes of the annual mean values of two parameters, chlorophyl-a and Secchi depth, are shown in Fig. 5 for the two

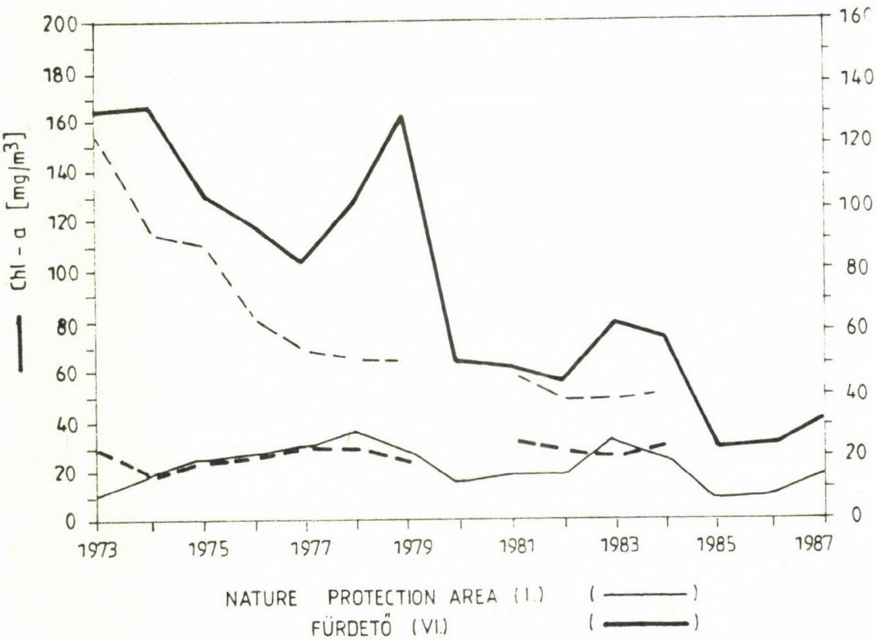


Figure 5: Annual mean values of chlorophyl-a and Secchi transparency in two areas

selected areas. While chlorophyl-a characterizes the trophic state, Secchi depth is a measure of resuspension that has been modified upon the effects of dredging.

In Fürdető there was a marked drop in the annual mean values of chlorophyl-a; from the 160 mg/m^3 value in 1973 to 30 mg/m^3 in 1985. Although the curve shows local maximum values (in 1979 and in 1983), the tendency of decreasing trophic levels is clear. Parallel with the decreasing algal concentrations the Secchi transparency has been slightly increasing.

In the nature preserve area (area I, in Fig.3) annual mean chlorophyl-a concentrations were relatively low (lower than 40 mg/m^3) in the period investigated. The tendency of changes, however, was very similar to that of Fürdető, only the rate of variation was smaller in terms of chlorophyl-a. In terms of Secchi transparency there was a monotonous decrease from 120 cm in 1973 to 40 cm in 1981. Since this area was not subjected to engineering control measures these changes are probably due to processes occurring in the surrounding areas (increased resuspension, decreased isolation can be mentioned as examples).

The areally weighted average chlorophyl-a concentration of the water of Lake Velence has been doubled (Fig.6) during the period of 1973-1978, all in accordance with the economic development that took place in the catchment basin. Between 1979 and 1985 chlorophyl-a values have decreased significantly, in spite of the continuing development in the catchment basin. The annual mean value dropped beyond 20 mg/m^3 being lower than that of 1973. The chemical oxygen demand of the lake water was decreased, in average, by approximately 4 g/m^3. This was partly due to the removal of soft bed material, containing many organic substances, and partly to the decrease in trophicity level. Due to increasing open water surfaces internal currents were induced and resuspension increased. Because of dredging down to the hard bottom material the lake's bottom surface became covered by colloidal substances that have bad settling properties.

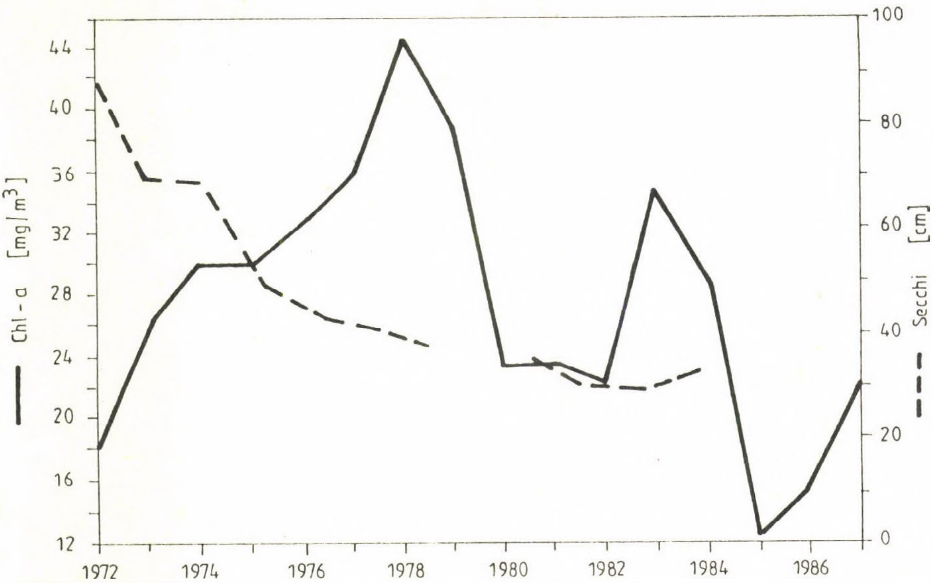


Figure 6: Changes of average annual chlorophyl-a concentration and Secchi transparency in Lake Velence (1972-1987, areally weighted yearly averages)

Due to higher resuspension and lowered settling velocities the transparency of the lake dropped to the third of the former value. This could have contributed, via shading effects, to the decrease in trophic levels.

It should be noted that the engineering control measures made in favour of recreational demands had partially unfavourable effects from the viewpoint of nature protection. On the basis of data on water chemistry, phytoplankton and zooplankton investigations it is apparent that the earlier mosaicness of patchiness of the Lake has been diminishing (i.e. the ecotones separating the former unique "clearings" - resource patches - of the lake have mostly deteriorated (Felföldy, 1977, Tóth 1980). Dredging has damaged the reed stock and the intensified wave action resulted in the spreading and faster growth of reed pests (Kovács et al. 1985). It shall be emphasized, however, that nature conservation interests have been given priority in the nature preserve area of the lake, where no engineering control measures have been implemented.

Summarizing it can be stated that due to the effects of engineering lake management strategies the trophic level of the lake has been decreased. Improvement in the trophic state affected the various lake regions differently, being eventually higher at the formerly more eutrophic eastern parts. In these latter regions the water quality became much better for meeting recreational water demands.

References

Akantisz, Zs. (1971): Improving the water quality of Lake Velence (hydraulic model study), (in Hungarian*). VITUKI Research Report, Budapest

Antalné Angster, M. (1987): Report on the water level regulation and water resource management of Lake Velence in 1986. KDT-VIZIG Research Report, Székesfehérvár

Baranyi, S. (1976): Summary evaluation of the regulation of Lake Velence and the relevant research results; 1971-1975. VITUKI Research Report, No: 7787/2/2, Budapest

Bendeffi, L. (1969): Formation and development of Lake Velence, VITUKI Research Report, No: 2503/C-B 40, Budapest

Felföldy, L. (1969): Research into Lake Velence in 1969. VITUKI Research Report No: 7784/4568, Budapest

Felföldy, L. (1971): Investigating the water quality of Lake Velence. VITUKI Research Report No: 7782/4/3, Budapest

Felföldy, L. (1972): Investigating the water quality of Lake Velence. VITUKI Research Report No: 7782/4/6, Budapest

Felföldy, L. (1974): Investigating the water quality of Lake Velence. VITUKI Research Report No: 7782/7/8, Budapest

Felföldy, L. (1975): Investigating the water quality of Lake Velence. VITUKI Research Report No: 7782/4/8, Budapest

* All references are in Hungarian except when otherwise indicated

Felföldy, L. (1977): Investigating the water quality of Lake Velence. VITUKI Research Report No: 7783/3/34, Budapest

Felföldy, L. (1979): Investigating the water quality and the environment of the nature preserve of Lake Velence. VITUKI Research Report No: 7783/s/222, Budapest

Jolánkai, G., Somlyódy, L. (1981): Nutrient loading estimate for Lake Balaton (original: English). Collaborative Paper No. CP-81-21, of the International Institute for Applied Systems Analysis, Laxenburg, Austria

Karászi, K. (editor) (1984): Recreation of Lake Velence. VIZDOK series VMGT No: 149,Budapest, ISBN 0324-2501.

Kovács, M., Gorzó, Gy., Gacsó, L., Busics, L. (1985): Survey and evaluation of the reed stock of Lake Velence. Report of the Geodetical and Cartographical Association,Budapest

Statistical Year Books (1956,1957...1986): KSH, Budapest

Szilágyi,F. (1982): Chemical analysis of the bottom sediment of Lake Velence. VITUKI Research Report No: 722/3/6, Budapest

Tóth, L. (1973): Qualitative and quantitative investigation of the fish stock of Lake Velence. VITUKI Research Report No: 7787/4/32, Budapest

Tóth, L. (1980): Research into the nature conservation and environmental protection aspects of the recreational region of Lake Velence. VITUKI Research Report No: 7783/3/199, Budapest

Tóth, L. (1981): Water quality investigation of Creek Vereb-Pázmándi. VITUKI Research Report No: 7783/3/290,Budapest

Tóth, L. (1983): Water quality investigation of Creek Vereb-Pázmándi. VITUKI Research Report No: 7613-3/3/40,Budapest

Tóth,L. (1985): The effects of engineering control strategies on the water quality of Lake Velence. VITUKI Research Report No: 7612/3/4, Budapest

Tóth, L. (1987): Elaboration of strategies for improving the water quality of Creek Császárviz. VITUKI Research Report No: 7622/3/11

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

CHARACTERISTICS OF TRIPTON SEDIMENTATION IN TEN SMALL LAKES

R. ŻYTKOWICZ

Department of Hydrobiology, N. Copernicus University
ul. Gagarina 9, Toruń, Poland

Sedimentation studies have become common in the last decade, due to the need of budget studies necessary for the rational water management.

This work presents the results obtained in a number of small and medium-sized lakes. The variety of the limnological types of these lakes, the strong and changeable external influences have caused that the observed sedimentation processes differ in quality and quantity from those known in big lakes. Because of their sensitivity to external influences, small lakes have to be managed with great care and they frequently require protective and restorative measures.

STUDY AREA AND METHODS

Sedimentation studies were undertaken in 38 lakes of different character. This paper presents the selected observations derived from 10 lakes where the most complete data have been obtained. Their general characteristics are shown in Table 1. The investigated basins belong to the Pomeranian and Masurian Lakelands (NW and NE Poland). The studies were carried out in 1975-1982. Here, the majority of data derives from 1977-1978.

Sedimentation rate was measured using the set of 3 sediment traps made of PCV tubes (internal ø 10 cm, height 30 cm), according to the recommendations of Blomqvist and Hakanson (1981). Sets of traps were placed in the deepest point of each lake at two levels: 1.5 m above the bottom and just below the thermocline. Traps were sampled at 2-4-week intervals, at least

Table 1. Characteristics of the investigated lakes

Lake	Surf. area (ha)	Mean depth (m)	Secchi disc range	pH (mean)	$P-PO_4$ index	N-min index	Other information
1. Czarne	9	4.2	2.5-7.8!	5.2	+	:	P,SW,LL,SE
2. Gacno	13	3.3	4.0-6.1!	5.6	+	:	P,SW,LL,SE
3. Jasne	11	8.1	2.8-12.2	5.2	:	.	D,SW,SE
4. Laska	70	1.5	1.5-2.5!	8.0	++	++	P,SR
5. Nawionek	11	6.3	3.6-7.6	5.9	:	:	D,SW,LL,SE
6. Skarlińskie	294	7.5	0.9-3.8	7.9	+	+	D
7. Suskie	63	2.3	0.2-1.8	8.0	+++	++++	P
8. Trzemeszno	184	2.1	0.3-0.9	7.9	++	++	P
9. Tynwałdzkie	32	2.0	0.4-1.4	7.9	++	+	P
10. Zmarłe	30	9.3	2.0-9.0	7.8	+	++	D,SE

Explanations: P = polymictic, D = dimictic, SW = soft-water, LL = Lobelia lake, SE = seepage, SR = short retention time, (!) = SD to the bottom, (+) = conc. = 0.2 $mg \cdot dm^{-3}$, (:) = conc. below 0.2 $mg \cdot dm^{-3}$, (.) = traces. Years of examination: No. 6 = 1976, No. 3 = 1978, others = 1977.

through one ice-free season. Big funnel traps of the shape of a quantitative plankton net (total height 150 cm, inlet ø 70 cm) were used additionally in special studies. The above traps, made of PCV film, were placed for 24 h every 2-4 weeks.

RESULTS AND DISCUSSION

Quantity and quality of sedimenting tripton

The measured sedimentation rate oscillated in a wide range in the studied basins (Table 2). A very low deposition was found in lakes Nos 1-3 and 5. These are soft-water, acid lakes of a high transparency and a low trophy level. Medium values were observed in more typical, eutrophic lakes Nos 6 and 10. In lakes Nos 4, 7, 8 and 9 (shallow, poly- and hypertrophic) an extremely high rate of sedimentation was measured.

Usually, sedimentation was measured during the ice-free season. Once it was evaluated under the ice cover in 15 basins of the same area (02.1980). To improve the sensitivity of

Table 2. General characteristics of tripton amount and quality

Lake	Sedimentation per day (mean)	Sedimentation per year (sum)	Organic matter content (%)	Energy content ($kJ \cdot g^{-1}$ org. matter)
	($g \cdot m^{-2}$ d.w.)			
1. Czarne	0.26	65	57	19.1
2. Gacno	0.24	60	59	19.4
3. Jasne	0.20	50	83	20.2
4. Laska	8.89	2222	42	18.4
5. Nawionek	0.32	80	66	20.4
6. Skarlińskie	2.04	510	63	20.3
7. Suskie	11.60	2900	59	24.6
8. Trzemeszno	4.76	1190	61	22.1
9. Tynwałdzkie	19.56	4890	54	20.3
10. Zmarłe	1.41	352	73	19.5

measurement, big funnel traps were placed for two weeks in the lakes. The sedimentation rate thus obtained was very low ranging from 0.003 to 0.286 $g \cdot m^{-2}$ day^{-1} (over the bottom), and it exceeded 0.1 $g \cdot m^{-2}$ d^{-1} in two lakes only. This accounts for the fact that the winter period was omitted in estimating the annual sedimentation.

Tripton contains both a very high percentage of organic matter and high energy (Table 2). A distinct reduction of these parameters is seen in lake No. 4 only. This correlates with the very high dynamism of its water.

Resuspension processes

The sedimentation picture in small or shallow basins is substantially influenced by resedimentation. This was not the case in Lobelia lakes Nos 1 and 2 (where weak, convectional mixing prevailed) due to stabilization of sediments by a coarse detritus and filamentous algal cover. In the four lakes (of similar trophy), showing the highest trap sedimentation, resuspension was much higher. Contrary to expectations, in the biggest and most shallow lakes Nos 4 and 8, sedimentation was the lowest. This was caused by submerged macrophytes covering a part of the bottom.

The attempt to estimate a real sedimentation in a strongly polymictic basin was undertaken in Lake Tynwałdzkie. Standard traps placed in lake enclosures (2x2 m, PCV foil) and big funnel traps collecting tripton from the isolated water column (foil 'sleeve' suspended from surface to the inlet of the funnel) were used. The mean rate thus obtained was 4.0 and 4.8 $g \cdot m^{-2}\ day^{-1}$, respectively. A similar value (4.5) was computed with the Gasith (1975) formula. The rough estimates were 4 times lower than the trap sedimentation in the open lake.

Under the ice cover, 3 to 11 times more tripton in the overbottom traps was observed in comparison to the traps at a depth of 6 m. Numerous migrating larvae of Chaoborus, Sergentia and specimens of P. quadrispinosa were found in the traps where most evident increase of sedimentation was seen.

Long-term variations

The sedimentation in Lake Zmarłe was measured from 1977 to 1980. The reduction of summer sedimentation from 1.41 to 0.26 $g \cdot m^{-2}\ day^{-1}$ took place at that time (Fig. 1). A similar tendency was observed in chemical indices (Giziński et al. 1989). It is ascribed to hydrological changes (gradual increase of precipitation). Big differences of sedimentation in two years, connected with fluctuations of primary production, were reported, e.g., by Anderson and Lastein (1981).

The other example of changes of sedimentation pattern in successive years is presented in Fig. 2. The drastic reduction of trap sedimentation was caused by restriction of sediment resuspension by covering it with an algal mat due to the growth of transparency (after a mass development of D. magna). The initial situation recurred after a few years.

Such rapid and great changes in the processes of sedimentation and resuspension induced both by auto- and allochthonous factors, can lead to deep abiotic and biocenotic modifications (e.g. strong trophic fluctuations).

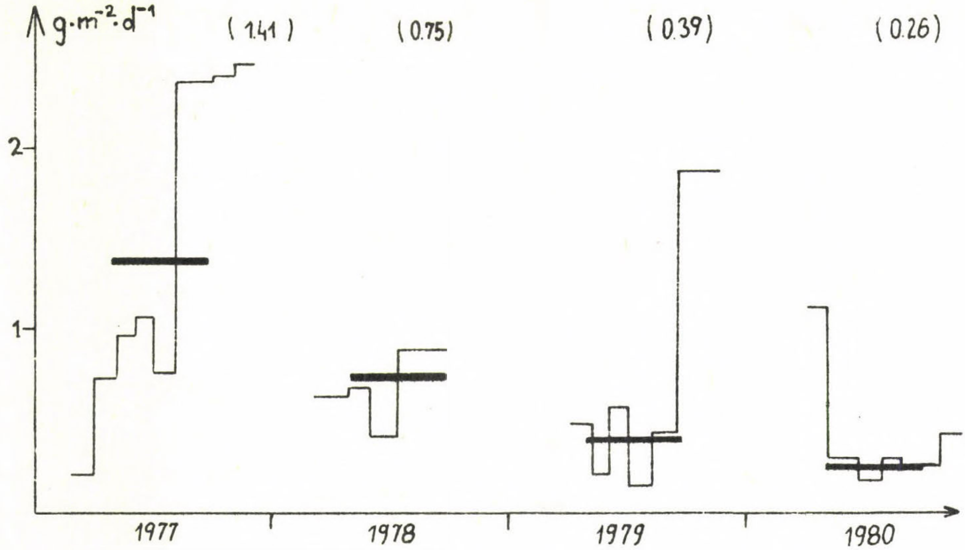


Fig. 1. Sedimentation rates in Lake Zmarłe, 1977-1980 (mean for the summer stratification period shown in parentheses and marked by horizontal lines).

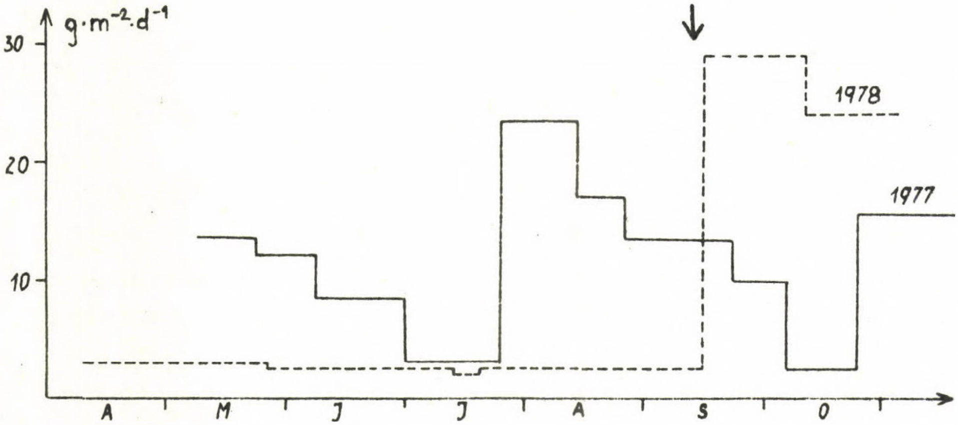


Fig. 2. Sedimentation pattern in two successive years in polymictic Lake Tynwałdzkie (solid line - 1977, mass abundance of blue-greens, Secchi disc 0.2-0.3 m; dotted line - 1978, mass development of D. magna; Secchi disc to the bottom; arrow indicates the time of algal mat collapse).

Horizontal variability

Most sedimentation measurements are carried out only at one site in small and regular lakes. To estimate horizontal differences, parallel measurements were taken at 8 points of Lake Zmarłe in 1979. All traps were distributed in the upper hypolimnion (10 m) 1.5 to 9 m above the bottom surface.

Table 3. Differentiation of sedimentation measured at 8 sites of Lake Zmarłe (in g d.w. m^{-2} day^{-1})

Date	5.5-30.5	30.5-21.6	21.6-1.8	1.8-17.9	17.9-19.11
Range	0.43-0.81	0.19-0.43	0.20-0.54	0.17-1.25	0.95-2.17
$\bar{x}\pm SD$	0.56 ± 0.14	0.31 ± 0.08	0.38 ± 0.12	0.74 ± 0.40	1.45 ± 0.43

As shown in Table 3 the essential differences were found in particular terms, whereas the annual mean sedimentation rates calculated for separate sites were much the same ($\bar{x}=0.82\pm0.10$).

REFERENCES

Anderson, F.Ø., Lastein, E. (1981): Sedimentation and resuspension in shallow eutrophic Lake Arreskov, Denmark. Verh. Internat. Verein. Limnol., 21, 425-430.

Blomqvist, S., Hakanson, L. (1981): A review on sediment traps in aquatic environments. Arch. Hydrobiol., 91, (1) 101-132.

Gasith, A. (1975): Tripton sedimentation in eutrophic lakes - simple correction for the resuspended matter. Verh. Internat. Verein. Limnol., 19, 116-122.

Giziński, A., Błedzki, L.A., Kentzer, A., Wiśniewski, R., Zawiślak, W., Żbikowski, J., Żytkowicz, R. (1989): Abiotic and zoocenotic conditions of ecosystem functioning in lakes of different trophy levels (in press).

LAKE MANAGEMENT AS AN ECOLOGICAL, ECONOMIC AND JURISDICTIONAL COMPLEX

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

THE GREAT LAKES MANAGEMENT EXPERIENCE AS A MODEL FOR THE SECOND ECOLOGICAL REVOLUTION

E. DOWDESWELL, V. ADAMKUS*

Ontario Environment, Canada
*U.S. Environmental Protection Agency, U.S.A.

Those interested in environmental policy have long understood the interdependent nature of the natural environment. From an ecological perspective everything is connected to everything else. Indeed, in the United States these interrelationships were a critical motivating factor in the establishment of the Environmental Protection Agency in 1970.

Even though ecological interdependence is a global phenomenon, most of our past actions to protect the environment have focussed on "local" problems. It was what one historian has described as the first ecological revolution: a popular recognition of the links between different aspects of the micro-environment. In the United States we have worked to protect particular water bodies and air sheds, and to clean up particular hazardous waste sites. And we have passed federal laws to restrict the use, storage and disposal of specific chemicals.

During these early years, the international dimension was not totally ignored. In the 1970's some bilateral efforts were made to exchange scientific or technical information or to solve problems along a common border. For example, the United States and Canada signed the Great Lakes Agreement in 1972 (and multi-national efforts were made to confront the ozone depletion issue). But from the perspectives of either scientific understanding or political co-operation, the awareness of international environmental issues remained in a very embryonic stage. Now in the late 1980's all that has changed and we stand at the threshold of the second ecological revolution. For a number of reasons, international environmental issues are now headline news. The Greenhouse effect, the loss of stratospheric ozone, the long range transport and deposition of toxics, and the worldwide effects of ocean pollution are topics of general widespread concern. The problem is that we have no international mechanisms or institutions to deal with problems that do not respect national borders...and offer no passports upon entry.

What we see today is a growing awareness that pollution and pollution control are problems whose solutions lie in international co-operation. It is this changing awareness that marks the beginning of the second ecological revolution...a recognition of the links in the macro-environment: that cutting the rain forests in South America may alter the climate in more temperate zones, that the pollution of air or water in one country can easily degrade the resources of another, that national treasures like Lake Balaton, and the Great Lakes are vulnerable to contamination from international sources...and will require multi-national co-operation for their management and preservation. Conferences like this one reflect the growing necessity to find co-opeative ways to protect shared national resources.

This second ecological revolution recognizes that damage to the global environment threatens the basic condition on which life depends and poses a clear and present danger that requires a global response. The emergence of an international environmental consciousness - this second ecological revolution - will continue to grow into the 21st century and has ramifications for all of us, whether we are scientists, environmentalists, government policy makers or simply individuals concerned about the overall quality of life shared on a very small planet spinning in space.

Resolving these kinds of global issues will be extraordinarily complex. Global environmental problems tend to be caused by total loadings of many different kinds of pollutants, most of which are emitted by the economically developed nations. As the global economy and global population expand, those total loadings will tend to increase and while the economically developed nations may be willing to reduce their per capita pollutant loadings in the interests of the global environment, the developing nations will be working very hard to expand their economies to catch up. Thus existing economic inequities - real or perceived - may make it very difficult for different nations with widely different cultural, political and economic systems to see that it is in their best national interests to work for the common global intereest in a kind of global partnership.

Fortunately, there are some models or "blueprints" which can guide us in this new area of environmental control. One of these is the Montreal protocol, another is the Great Lakes Water Quality Agreement.

For example, let me give you just a brief overview of what the agreements have resulted in on the U.S. side. Since enactment of the federal Water Pollution Control Act of 1972, 4,800 permits for industrial and municipal sewage treatment works have been issued to control discharges into the waters of the Great Lakes system. In order to help municipalities come into compliance with their permit requirements, we have spent $7.9 billion in federal and state grants. As a result of this investment, today we see an 80% reduction in phosphorus loadings, and similar reductions in biological oxygen demand and suspended solids. Despite these successes, the job is far from over. But what is important is that we have developed and implemented an international mechanism to deal with a common environmental problem...and it works.

The Montreal Protocol, signed last September, is also an exemplary model of a global compromise that subjugated national interests to the more global common good. The Agreement is designed to curtail the production and use of chlorofluorocarbons and halons worldwide. Two things are noteworthy. First, the treaty proves that national governments are capable of agreeing on an environmental protection program <u>before</u> major health or environmental problems occur. Secondly, it demonstrates the importance of developing a scientific consensus before attempting to reach political consensus.

What makes the Montreal Protocol an important model for international co-operation is that it recognizes up front the differences of the various parties and then seeks innovative solutions to those differences.

With tools like these and an awareness of the global impacts of environmental problems, I hold great hope for the future.

In international meetings like this one, scientists are sharing the kind of information needed to build an international political consensus. Government officials are beginning to discuss the kinds of consistent environmental regulations needed. Bilateral and multi-lateral agreements are being negotiated.

But that is not enough. Rhetoric cannot clean up a river and good intentions cannot clear the air. Further, we know that the earth's ecological systems do not respond to piecemeal improvements. If we believe that environmental problems are truly global problems, then solutions too must be implemented on a global scale.

But when the rhetoric stops, we must come face to face with some very difficult questions.

Simply put, we must ask how such a massive undertaking is to be financed and carried out. For unless we find a means to put the necessary technology into the hands of those who need it, the dream of global environmental protection will die.

I propose that the costs of environmental clean-up and protection be borne in common by the nations of the world based on the ability of each to pay. Clearly this means that the larger developed nations will bear a greater financial burden in this effort. But do we really have an option? We cannot wait for each nation to be able to afford the costs of pollution control. The risks I believe are too high.

I believe that an international pollution control strategy, backed with an international financial commitment to put the necessary technology in place can succeed.

In Canada and the U.S. we have been trying to meet the challenge. This past year we have renegotiated a "new and improved" Great Lakes Water Quality Agreement.

As well, through the International Joint Commission, the two countries have begun a study to "examine and report upon methods of alleviating the adverse consequences of fluctuating water levels in the Great Lakes-St. Lawrence River Basin". The Reference appears simple and straightforward, but the IJC is enthusiastically portraying it as a major, comprehensive reference, unprecedented in scope, broader in scale, requiring new approaches, innovation and creativity.

Indeed, an exploration of the full range of possible factors involved in fluctuating lake levels of necessity involves looking at:

- climate change;
- land use and shoreline management;
- costs and benefits of lake regulation schemes;
- engineering solutions; and
- environmental, economic, social and political impacts.

We must examine effects within and outside the Basin, and in differing time dimensions - ranging from emergency measures to long term consequences. Add to this the complexity of the system - geographically, hydraulically, politically - and you have some idea of the nature of the challenge.

In each of these two activities- we've learned something about our approach to lake management - and perhaps also the management of other global environmental problems. Not surprisingly the lessons are very similar to some we heard about yesterday. I've chosen three to comment on today.

First we've learned (once again) that knowledge and knowledge-building must be the underpinning of any agreement or study associated with the rational management of water systems. Consequently, research is explicity cited in the specific provisions of the revised Agreement. This conference need not be reminded of how the seminal work of Dr. Vollenweider and his colleagues on the eutrophication of lakes provided the basis for the establishment of phosphorous reduction targets in the original 1972 Great Lakes Agreement. The success of this action has meant that we've now moved on to other challenges - for example, toxic chemicals.

While we are still plagued by the inability to precisely attribute significance to the detection of minute quantities of toxic chemicals in the environment, our knowledge on impacts and pathways of toxics has increased. For instance, today we are actively involved in developing ecosystem health indicators for the lakes based not on specific concentrations of contaminants in water but on other parameters such as the abundance and health of a specific species of fish, waterfowl or other forms of aquatic life. That some degree of scientific consensus has been achieved on some ecosystem health indicators is a definite sign of the progress of knowledge-building. The advances in our knowledge of toxic chemicals in the Great Lakes developed to the point that the new Agreement has specific annexes on contaminated sediments, groundwater, airborne toxics and non-point sources. Our policy instrument - the Agreement - continues to be modified as our knowledge evolves.

But we've also discovered gaps in our scientific knowledge and understanding. Although we are very ready to use the term "ecosystem"approach", we have a lot to learn about how to take a broad systemic view of the interaction among the physical, biological and chemical components in the Great Lakes Basin; how to take a geographically comprehensive view - looking at land and air and water; and, how to include <u>humans</u> as an integral factor in the well-being of the system (recognizing the social, economic, political and technical variables that affect how humans use and live with natural resources).

We don't yet know enough about conflicting human value systems, uncertainty and risk, maximum stress conditions, limits to technology, socioeconomic tradeoffs and sustainable development.

But, as you well know, knowledge and knowledge-building are dynamic entities. As our knowledge develops, our understanding and sensitivity of the world environment will force us to act.

The second thing that has been reinforced by these two activities is the need for strong yet flexible institutional arrangements. Neither Canada nor the United States, acting independently of one another, would be able to effect restorative or preventative measures for the Lakes which we share.

Too often the laws that are passed and the regulatory frameworks that are developed only affect pollution problems within the bounds of an existing nation-state. They don't deal with interjurisdictional impacts.

Canada and the U.S. are fortunate in having a unique bilateral institution called the International Joint Commission through which cooperative systematic study and recommendations for control actions can be carried out and developed. The Commission was established in 1909 under the Boundary Waters Treaty between Canada and the U.S. to ensure the equitable uses of the boundary waters between our two nations. The IJC is a permanent body and it receives "references" from both countries to undertake specific activities. In 1964 it was requested to determine the nature and extent of pollution of the Great Lakes and recommend remedial measures. Its recommendations to set basic water quality objectives for the lakes, to develop a range of pollution control programs to achieve and maintain those objectives, and to report continually to the public and the governments on progress of implementing the preceeding were accepted by governments and led to the first Great Lakes Water Quality Agreement.

The basic structure of the Agreement remains the same today -- although it evolved in 1978 to include strengthened toxic controls -- and in 1987 to include the development of ecosystem objectives, site specific remedial action plans and lake wide management plans. By maintaining its focus on the environmental imperatives of the Lakes, the Agreement has forced governments to find ways to continue to be responsive.

Governments have purposely committed themselves to be evaluated by the IJC (which reports directly to the public on the state of the lakes and the adequacy of government programs). This commitment to communication provides the necessary public accountability component to the overall Great Lakes management process.

This institutional arrangement - two responsible parties and an independent third party evaluator -- is unique. I think it has potential for wider application to other multi-jurisdictional resource management settings.

And finally -- the third thing that we've learned is that we have to be open to new approaches. It's understandable that we tend to cling to old but familiar ways of doing things. It may be understandable -- but no longer appropriate. We have to expand and rethink the way things have been done in the past -- raise new questions, new possibilities and regard problems from a new angle.

Perhaps scientists have always been able to do that. But environmental managers have not. There comes a time when you can't solve problems with existing tools -- in the old context. We become trapped in the language and questions of the past. Instead of asking -- how can we mitigate the damages of fluctuating lake levels, perhaps the question should be -- how can we help man adapt?

We need to try harder to achieve an integrated whole system approach to the management of problems in the Great Lakes. Multidisciplinary studies alone are not enough. The specialist biases and political conflicts that come with a segmented approach inhibit innovation. We need real synergy. It is much as a symphony requiring a conductor skilled at blending and integrating -- at just the right time -- in the right proportion.

At the same time, we need to accept divergent thinking. And one of the areas in which our thinking needs to change has to do with the role of the public.

Today we have a highly sophisticated and educated public whose knowledge base is high and whose level of concern for environmental quality is strong. Certainly this is the case in the Great Lakes Basin. We have several hundred separate citizen groups. They started out as local single-issue groups whose focus and concern has now broadened to the entire Great Lakes. They have full-time professional staff keeping them well informed.

One example of their commitment and concern is found in a remarkable project carried out by Great Lakes United (a bilateral federation of over 200 environmental groups). Through private funding from foundations they held 19 public hearings all around the Great Lakes on the effectiveness of the Great Lakes Water Quality Agreement. As we renegotiated that Agreement you can be sure their views were considered.

Involving the public is administratively untidy -- and somewhat risky. Any public process creates expectations from which one cannot back down. But the public must be considered a source of good knowledge in its own right -- and, ultimately the public has to be a part of the solution to environmental problems. Clearly the potential rewards outweigh the risk.

We're taking that risk. In our newest initiatives under the Agreement, Canada and the U.S. have agreed to develop and implement Remedial Action Plans for 42 severely degraded areas around the lakes. We are commited to an intensive consultation program with citizens advisory committees working with government scientific and technical experts to specifically define the environmental problem, select remedial options and monitor their implementation in accordance with a negotiated schedule.

The process, needless to say, is arduous and unpredictable -- but we remain confident that the results will warrant the effort.

So, to come back to the central theme. As we tackle the complex environmental issues of the 80's and 90's and as our knowledge of environmental problems on a global basis grows, it is inevitable that effective multijurisdictional action is the only response.

We have models. The Great Lakes management experience has taught us something about the importance of our knowledge base, about the strength of our cooperative institutional arrangements, and about the need for flexible mind-sets in our management approaches. It's an experience of pragmatism and vision.

I invite you to study our experience as collectively we seek to improve the way in which we address global environmental challenges -- in this second ecological revolution.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

LAKEWIDE MANAGEMENT PLANS: A MULTI-JURISDICTIONAL APPROACH TO ECOSYSTEM RESTORATION

C. FINCH, R. SHIMIZU*

Great Lakes National Program Office, U.S. Environmental Protection Agency
230 South Dearborn Avenue, Chicago, IL 60604, U.S.A.
*Great Lakes Environmental Program Office, Environment Canada, 25 St. Clair Avenue East, Toronto M4T 1M2, Canada

ABSTRACT

The Great Lakes Water Quality Agreement represents a joint commitment between the governments of the United States of America and Canada to restore and enhance water quality in the Great Lakes system. Originally signed in 1972, the Agreement was amended by Protocol late in 1987 and expanded to incorporate the development and implementation of lakewide management plans. This recognized a need to focus attention on ambient water quality problems that continued to adversely impact the ecosystem health of the Great Lakes despite reductions that had been accomplished through regulatory controls. The lakewide management plans embody a systematic and comprehensive ecosystem approach to pollution abatement, focusing on toxic contaminants originating from sources not yet subject to adequate control. Specific activities associated with lakewide management plans include measurement of toxic substances in all major components of the ecosystem, together with development and application of source, fate and transport models, uniform and more stringent criteria and standards and, ultimately, ecosystem-based objectives. To date, the Parties have undertaken several activities that help set direction for full implementation of lakewide management plans. This work has been carried out in the true context of multijurisdictional management conducted in a climate of wide public consultation.

INTRODUCTION

In 1972, the governments of the United States of America and Canada signed the Great Lakes Water Quality Agreement, an international accord that established a basis for cooperative management of pollution issues in the Great Lakes Basin. This has proven to be a remarkably effective endeavour:

(i) accelerated eutrophication has been checked through the development and imposition of stringent phosphorus controls;

(ii) treatment of sewage discharges throughout the Basin has been substantially upgraded;

(iii) point source discharges have been controlled to the extent that levels of many persistent toxic chemicals in both water and biota have been reduced.

Over time, the Agreement has been expanded and modified to respond to emerging issues, the most recent of which is the insidious and pervasive contamination of the Great Lakes Basin by toxic substances. As the Agreement has evolved, it has taken on certain unique features which offer a framework for systematic multijurisdictional management of the lakes from a more holistic perspective. These features have derived from the indisputable premise that pollution is a highly complex phenomenon having utter disregard for man-made boundaries. As such, they will likely find application elsewhere. One such feature, the lakewide management plan, was recently incorporated into the Agreement and will be subject of this talk.

LAKEWIDE MANAGEMENT PLANS

The concept of lakewide management plans originated with the realization that while there had been a measurable improvement in lake conditions to regulatory controls for toxic substances, levels of these materials in sediments and biota were still cause for concern. It became evident that there were many more sources of chemical contamination to the Lakes than the industrial inputs which had been subjected to regulations. Continued ecosystem health problems within the Great Lakes Basin are manifested in terms of, for example, restricted fish consumption and wildlife deformities. There is a growing suspicion that the "contaminant" burden of the Great Lakes ecosystem is adversely affecting all inhabitants of the Basin. The lakewide management plan as provided under the Agreement is a tool for use by the Parties to systematically address from a holistic perspective the ongoing pollution from toxic chemicals.

The thrust and intent of a lakewide management plan is the scheduled reduction of loadings of Critical Pollutants in order to recover and maintain the integrity of the lake in terms of its ability to support all beneficial uses. Succinctly, Critical Pollutants have been defined, under the Agreement, as substances persisting at levels which singly or in combination threaten this integrity.

EVOLUTION OF LAKEWIDE MANAGEMENT PLANS

Lakewide management plans were formally incorporated into the Agreement in the Fall of 1987 with the signing of an amending Protocol. Prior to this the Parties, both unilaterally and bilaterally, had initiated efforts towards establishing the concept of comprehensive and systematic lakewide management. Such efforts will be reviewed briefly.

1. Lake Michigan

The Lake Michigan Toxic Pollutant Control/Reduction Strategy was prepared as an implementation plan to eliminate toxic pollutant problems from the Lake Michigan system. This is a multijurisdictional effort involving the United States Environmental Protection Agency, the Indiana Department of Environmental Management, the Michigan Department of Natural Resources and the Wisconsin Department of Natural Resources. The Strategy is comprised of two phases. The first contains a number of discrete but inter-related elements intended to define, quantify and control the major toxicant problems in Lake Michigan. This involves developing a list of pollutants of concern, establishing appropriate effluent limits for these materials, and constructing a mass balance to model sources, fate and transport. Should Phase I show that established controls are insufficient to achieve desired lake restoration, a second phase of scheduled total lake load-reduction plans will be implemented.

Since Phase I will have addressed major point source contributions to Lake Michigan, the effectiveness of additional load reductions in Phase II will be limited by jurisdictional authority over other identified inputs.

2. Niagara River

The Niagara River Toxics Management Plan is an example of a bilateral activity for toxic contamination management in a major boundary water course connecting Lake Erie to Lake Ontario. The Agreement, signed by representatives of the governments of the United States of America and Canada, gave rise to an exercise that contained a number of provisions. Under one the Parties agreed to reduce by 50% the loading of scheduled toxic chemicals into the Niagara River by 1996. This is to be accomplished by the designation of specific chemicals of concern which the Parties will have completed this fall. Also a systematic river monitoring program is conducted which provides an annual overview of the state of the river.

Another, more significant tenet in terms of lakewide management plans, committed the Parties to developing a similar toxic contaminant management strategy for Lake Ontario. It must be emphasized that both the commitment and the activity, as with the Lake Michigan and Green Bay programs, predated the 1987 Protocol to the Great Lakes Water Quality Agreement.

3. Lake Ontario

The Lake Ontario Toxics Management Plan was developed by the environmental agencies of the four major government jurisdictions around the Lake Ontario Basin. These are the United States Environmental Protection Agency, Environment Canada, the Ontario Ministry of the Environment and the New York State Department of Environmental Conservation.

There is one key point implicit to and underlying this entire discussion. A truly major accomplishment that is applicable both to the Great Lakes Water Quality Agreement as well as to these other activities is the

(a) establishment of uniform criteria and standards for maximum concentrations of substances in various media between the participating agencies;

(b) development and implementation of ecosystem objectives as the measure of whole lake restoration of aquatic integrity.

MASS BALANCE APPROACH TO MANAGEMENT OF TOXIC CHEMICALS

Traditionally, the management of water quality has focused on control of direct releases of pollutants. Such sources were the easiest to identify, characterize, and control. The regulatory laws to control sources of pollutants are media-specific, with specific laws dealing separately with air, water, and land pollution. For these reasons, restoration and maintenance of water quality were largely tied to control of point sources from which contaminants were discharged directly into the nation's waterways.

Recognition that pollutants are also indirectly introduced has led to reassessment of the traditional approach to management of Great Lakes water quality. Great Lakes water quality managers have concluded that adequate management of contaminants requires that the total contributions to pollution from all media and all types of sources be quantified, and that a mass balance approach, which allows for evaluation of the relative significance of multiple sources, be used for total load management.

Mass balancing has been successfully applied to the regulation of nutrient loads in the Great Lakes during the past decade; the current concern over toxic substances in the lakes signals the need for a similar approach to regulation of toxic substances. The sources, pathways, and sinks for toxics, however, are less well understood. It is, therefore, desirable to pilot the mass balance approach for toxics in a smaller ecosystem prior to expansion to all the Great Lakes. The Green Bay/Fox River Mass Balance Study has been designed to serve as such a pilot project.

In a mass balance the quantities of contaminants entering the system, minus the quantities stored, transformed, or degraded within the system, must equal the quantity leaving the system. The basic mass balance equation, based on the law of the conservation of mass, is presented below:

INPUT	+	GENERATION	-	OUTPUT	-	CONSUMPTION	=	ACCUMULATION
(enters through system boundaries)		(produced within system)		(leave through system boundaries)		(consumed within system)		(buildup within system)

An ecosystem can be portrayed as a series of boxes or compartments (sediments, water, biota, atmosphere, etc.) linked by arrows representing transfer processes. In its simplest form, a mass balance equation can be constructed by measuring the quantities of a contaminant entering the system, the quantities leaving the system, and the quantities present in the system (sediment, water, and biota compartments). The resulting equation, if balanced, provides a quantitative description of the movement of the contaminant through the system. If it is not balanced, it indicates that better understanding of the system dynamics and/or more accurate measurements are required in order to accurately describe the system dynamics.

The ultimate utility of the mass balance approach as a management tool is to prioritize and allocate resources for research, remedial actions, and regulatory efforts. Its usefulness depends on the ability to predict impacts of various management actions on one or more target compartments, e.g., contaminant levels in fish. Mathematical models that describe the interactions among compartments are thus an important component of the mass balance approach.

The Great Lakes have historically served as a proving ground for exploring solutions to environmental problems. The Great Lakes are deep and have a long retention time; in addition they have been exposed to a wide range of contaminants from industry, agriculture, and municipalities. Consequently, toxic contamination of fish, bioaccumulation of metals, and eutrophication are readily exhibited. Since the lakes are so sensitive to contamination, it has often been incumbent upon the state, provincial and federal environmental agencies within the Great Lakes Basin to develop and test innovative solution to environmental problems.

The concept of total load management in the Great Lakes Basin is a fundamental element of the Water Quality Agreement between Canada and the United States, of the Lake Ontario Toxics Management Plan, and of the Lake Michigan Toxicant Control Strategy. Great Lakes managers have recognized that addressing toxic contaminants in the Great Lakes system requires a comprehensive multi-media evaluation of point and nonpoint source loadings to the lakes. This requires going beyond the relatively simple consideration of point and land-based nonpoint source loadings that was used to determine phosphorus loading limits in addressing eutrophication problems in the 1970s. Current sources of persistent toxics are less likely to be point sources; rather, what needs to be determined is the extent to which there are significant reservoirs of persistent toxic substances in less easily measured media such as air, precipitation, soil, sediments, and groundwater.

The mass balance approach provides a tool with which managers can determine the relative amounts of persistent toxic substances that the various sources contribute to the environment, so that they can determine which environmental control programs should receive greater emphasis. Knowledge about the relative contributions of the different sources of contaminants and the relative costs of their removal or control can lead to more cost-effective approaches for remediation.

There is general agreement among the scientific community that existing ecosystem mass balance models are rudimentary and require further development in order to meet management needs. Great Lakes studies provide opportunities for testing existing models and developing more sophisticated models. Equally important, the studies promote the development of the improved environmental measurement technologies required for a mass balance approach for toxics.

Last but not least, the mass balance approach provides a valuable framework for the coordination of research activities among the various State and Federal agencies responsible for the protection and greater understanding of the environment. It is a framework for the coordination of environmental research within a given ecosystem by organizations with differing objectives. The information resulting from such a coordinated study is likely to allow conclusions to be drawn that could not have been drawn from any single project by itself.

The overall goal of the current Green Bay/Fox River Study is therefore to develop a modeling framework to improve our understanding of the sources, transport, and fate of toxic compounds, to evaluate the technological capability to measure multi-media loadings to a system, and ultimately to guide and support regulatory activity. The study will thus serve as a pilot for possible future modeling studies of Great Lakes ecosystems.

CONCLUDING REMARKS

The Lake Michigan Toxic Pollutant Control Strategy, the Lake Ontario Toxics Management Plan, and the Green Bay/Fox River Mass Balance study were all developed prior to, or concurrently with but uninfluenced by, the 1987 protocol amending the Great Lakes Water Quality Agreement. It is interesting to note, however, that although they were developed independently, they all contain elements called for in the Agreement, namely the identification of critical pollutants, determination of all sources for these pollutants and their transport and fate within the ecosystem, and finally efforts to develop total load management systems for the pollutants. The Lake Ontario Toxics Management Plan is the most comprehensive of all the activities although it does not follow in linear fashion the tenets of generic lakewide management plans. It does, however, contain most of the provisions specified for a lakewide management plan.

One difference between the Lake Ontario Plan and the generic definition of lakewide management plans lies in the scope. This is somewhat narrower in the Agreement in that Critical Pollutants are those substances which, despite the application of regulatory control, continue to be a problem in the Lake. Conversely, as the name of the Lake Ontario Plan suggests, its focus is on all toxic chemicals. The scheme for categorizing chemical contaminants and taking appropriate action, described under the Lake Ontario Plan, does provide the mechanism for rationalizing these two positions. The Agreement also incorporates an evaluation mechanism through a referee, the International Joint Commission. This provides the Parties and the general public with a systematically derived "report card" describing the success or failure of the Parties to implement Agreement requirements and the response of the lakes to the various programs and measures. The Lake Ontario Plan, on the other hand, provides for periodic reporting in a public forum but this only guarantees public scrutiny, not peer review. This inconsistency will likely be resolved in the near future as the policy arms of the signatory agencies have already agreed the Lake Ontario Plan will serve as the lakewide management plan required under the Agreement. It may be modified as experience and consultation with the International Joint Commission refine the generic requirements.

Of major importance in all these activities is the growing recognition that successful development of Lakewide Management Plans further depends on two essential elements. The first is that its involved jurisdictions develop uniform criteria and standards so that all involved reach a common understanding of what the problem contaminants are and what degree of restoration is desired. The second is that development of the plan incorporates full participation by all interested parties, including the public, local governments, industry, and regulatory agencies. It is only through this participation that the broad base of interest and support can be achieved which will lead to implementation of the resulting plan.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

LEARNING FROM NORTH AMERICAN GREAT LAKES DEGRADATION: THOROUGH ENVIRONMENTAL IMPACT ASSESSMENTS POINT THE WAY TO POLLUTION PREVENTION

T. DOBSON

109 Olds Hall, Michigan State University
E. Lansing, MI, U.S.A.

I. THE NORTH AMERICAN GREAT LAKES: A VALUABLE RESOURCE THREATENED

The North American Great Lakes -- Superior, Michigan, Huron, Erie, and Ontario -- comprise the largest freshwater system on Earth. Together they hold one-fifth of the planet's freshwater, with a surface area of approximately 245,299 km^2. The largest of the five, Superior, is 350 miles by 160 miles (82,000 km^2), measures 406 meters at its deepest point, and contains 2,935 cubic miles of water.* The smallest, Erie, by virtue of its shallow character, contains 116 cubic miles of water. In addition, the Lakes act as drainage for hundreds of tributary rivers located in the Great Lakes Basin.

Forty-four million Canadians and Americans live in the Great Lakes Basin and rely on the lakes to satisfy many of their basic water needs. In addition to meeting critical bathing and drinking needs, the lakes support a plethora of commercial activities. Twenty-percent of U.S. industry is located on the Great Lakes. Every year, thousands of cargo ships cross the lakes, moving goods from as far away as the Mississippi River, flowing through the country's midsection, all the way to the Atlantic. Commercial fishing also plays a significant role in the economy of the lakes region. With 67,000 of 201,000 square miles in the basin in agricultural land, agriculture is another major factor. Also crucial to the lakes' economies are the myriad recreational uses of the lakes, including sport fishing.

---

* By way of comparison, the surface area of Africa's Lake Victoria is approximately 75,000 km^2, that of Japan's Lake Biwa is 681 km^2, and that of Hungary's Lake Balaton is 596 km^2.

Canadians and Americans have long worked together to regulate and jointly accommodate use of this shared resource. In 1909, the two governments signed the Boundary Waters Treaty* in which they pledged not to harm the lakes. A binational agency, the International Joint Commission (IJC), was created by the Treaty,** and it continues to work actively to assist the two countries to preserve and improve water quality. It provides a model of international environmental problem-solving. In 1972, heightened concern about environmental protection led to the drafting and signing of the Great Lakes Water Quality Agreement*** that recognized the urgent need to address pollution problems in the lakes. The commitment to improving water quality was strengthened in 1978**** with the adoption of more stringent water quality objectives, a focus shifting to toxic contaminants, and an ecosystem approach. 1987 amendments***** seek to bring about improved management and accountability to make possible the attainment of the goals established in the earlier agreements.

As environmental awareness grew in the 1950's, 1960's, and 1970's, investigations of Great Lakes water quality yielded startling and troubling results. Most pressing among the problems uncovered was the condition of Erie, the smallest of the five lakes. It was declared to be dying, as it was in an advanced state of eutrophication. Happily, Erie has been pulled back from the brink of disaster, but those who live in the Great Lakes Basin have become accustomed to repeated and frequent reports of damage to the lakes and their ecosystems. For example, fish advisories, warning against consumption of certain species of fish, are common. Over 400 toxic chemicals have been found in the lakes. Even Isle Royale, an undeveloped island in northern Lake Superior, thought far beyond the reaches of pollution, was recently discovered to have significant levels of environmental contamination.+

---

* Boundary Waters Treaty, 36 Stat 2448-2455 (1909).

** Boundary Waters Treaty, 36 Stat 2451 (1909).

***Great Lakes Water Quality Agreement, 23 UST 301-369 (1972).

**** Great Lakes Water Quality Agreement, 30 UST 1383-1487 (1978).

***** 1987 amendments the Great Lakes Water Quality Agreement of 1978, __ UST __ (1987).

+ Carothers, A Desert of Waters: Toxic Pollution and the Great Lakes, Greenpeace 11 (July/Aug. 1988).

In seeking approaches to address environmental concerns in the Great Lakes Basin, the IJC has identified 42 Areas of Concern (AOCs) where damage is sufficiently severe or the area sufficiently sensitive to require remediation.

Examples taken from the list of 42 AOCs illustrate the types of sources of pollutants and the range of problems that pollution has caused.* Kimberly-Clark of Canada Limited operates a kraft pulp mill which discharges its effluent into the Blackbird Creek inlet of Jackfish Bay in Northern Lake Superior. In 1969 the waters of Jackfish Bay were observed to be turbid, malodorous, laden with suspended materials, and supporting high bacterial populations. In addition, stream bottom sediments were covered with wood wastes. Studies conducted in 1981 revealed significant levels of chromium, iron, copper, oil, grease, phosphorous, Kjeldahl nitrogen, zinc, cadmium, and mercury that exceeded IJC guidelines. Aquatic life had virtually disappeared from the creek and the small lakes that form a part of the creek system. The condition of the bay has created problems for both sport and commercial fisheries and recreational interests.**

Industrial and non-point sources seem to be the primary sources of pollution in the Saginaw River and Bay area in Eastern Lake Huron. High levels of PBB, DDT, and Tris have contaminated the sediments and fish in bodies of water in this area. Also noted with concern is the presence of hexachlorobenzene, polychlorinated dibenzofurans, dibenzo-p-dioxins, diphenyl ethers, styrenes and terphenyls, suspected to originate from Dow Chemical Company operations on the Tittabawassee River. High concentrations of PCBs originating from Cast Forge Company operations have resulted in fish consumption advisories. The presence of toxic

---

* A recent report by Greenpeace is considerably less sanguine than the IJC regarding the condition of the American Great Lakes. In "A Desert of Waters: Toxic Pollution and the Great Lakes," Andre Carothers describes them as a "750-mile-wide chain of industrial sewers." 13 Greenpeace No. 4, at 10, July/Aug (1988).

** Great Lakes Water Quality Bd., supra note 12, at 9-10.

contaminants is suspected to be effecting the reproductive success of fish-eating avian populations. Even though Michigan banned phosphorous detergents in 1977, phosphorous loading continues to be a problem in this area from fertilizer runoff and other non-point sources.*

The United States and Canada and local governments are working together to gather further data and to formulate clean-up plans.** The full costs of remediation and pollution prevention, while not known at the present time, will doubtless be staggering. For example, to clean up the Milwaukee Harbor in Wisconsin, $300 million in federal funds and $200 in state funds have been spent so far to construct a waste treatment facility. It is estimated that an additional $262 million in federal funds and $240 million in state funds will be required to complete the project by 1992.*** A control order proposed to reduce discharge of toxic wastes from Courtaulds, in the Cornwall-Lake St. Francis area of Ontario, would require that the company construct a treatment plant.**** Phosphorous reduction efforts in the basin between 1972 and 1982 are estimated to have cost $10 billion.*****

The costs in loss of commercial fishing, recreation, and harm to human health and the ecosystem have also caused significant losses to the economy. Not as readily quantifiable as direct costs, indirect costs include death, disability, lost work days, lost income and resultant taxes and increased medical care expenditures. A study of human health effects is underway in one AOC as a part of a Remedial Action Plan,****** but at the present time little is known of the extent and cost of these effects.******* Yet, as one weighs the costs and benefits of new

---

* Id. at 73-75.

** Cooperatively working to resolve these environmental problems is a daunting task given the number of governmental jurisdictions involved. P. Muldoon & M. Valiante, ZERO DISCHARGE; A STRAGTEGY FOR THE REGULATION OF TOXIC SUBSTANCES IN THE GREAT LAKES ECOSYSTEM 64-65 (1988).

*** Great Lakes Quality Bd., supra note 12, at 50.

**** Id. at 195.

***** P. Muldoon & M. Valiante, supra note 23, at 16.

****** E.g., AOC #10, Fox River/Southern Green Bay, Great Lakes Water Quality Bd., supra note 12, at 37.

******* Some investigations of human health effects have been conducted in Ontario. P. Muldoon & M. Valiante, supra note 23, at 14-15.

technology and development, such costs should be taken into account even in the absence of hard data regarding their magnitude. Billions of dollars must be spent for corrective measures, billions that could otherwise be put to productive use, creating jobs, products, and services. The currently calculable costs may represent but the tip of a large iceberg and provide a compelling argument for a re-evaluation of the economic development process.

II. TRACKING THE SOURCES OF ENVIRONMENTAL INJURY: THE COSTS OF TECHNOLOGY.

In order to begin to re-evaluate economic development processes, the facts must first be gathered, including pinpointing the sources of pollution. In identifying the specific sources of environmental damage in the Great Lakes Basin, it can be seen that a wide variety of technologies, employed to increase production, improve the quality of life, and boost the economy, have had the unfortunate additional effect of degrading the environment.

Technologies prevalent in the basin include those used in manufacturing, mining, and sewage treatment. These technologies were selected and put in place without critical analysis, mostly in an era when the environmental consequences were not known, understood, or even considered. Proponents envisioned the benefits to be gained but not the social costs. Inevitably, many of these technlogical processes created huge waste streams, often highly toxic, that were piped directly into the lakes or their tributaries.

We now need to engage in a retrospective critical evaluation of these technologies to learn how best to protect the fragile ecosystems upon which all life depends while maintaining a stable economy in the developed world and a growing and expanding economy in the less developed world. An example serves to illustrate our dilemma. In our rush to stimulate economic activity and enjoy the most modern equipment, we supported the creation of seemingly boundless supplies of electricity to fuel, among other things, manufacturing operations which provide goods and jobs. As a consequence of electric power generation, however, we also have acid rain, nuclear accidents, and nuclear waste.

Our retrospective evaluation should be informed by the thinking of scholars of philosophy of science who have critically analyzed technologies' multitudinous impacts on society, suggest that humankind creates technologies and then begins a process of adaptation to accommodate to the demands of the new technology ("reverse adaptation"*) rather than adapting technologies to existing human and ecosystem needs and systems. At the same time, we accept their "side effects" (so designated because, while they are clearly "effects" of technology, they are unintended and usually unwanted and so receive a different designation) as a necessary evil.**

In the area of water management, we have sacrificed small bodies of water such as creeks or rivers to pollution in order to gain the benefits of development. We now recognize, however, that the sacrifices were much more extensive than realized at the time it was decided to make them. Polluting water supplies for commercial gain was based on a series of miscalculations. As scientific research has discovered, contaminated water moves under and across land and through the air, carrying toxins to distant destinations where new problems are created. The growing realization that these problems exist leads inexorably to the conclusion that using water bodies as waste disposal sites was not the easy answer that it appeared to be.

To find more appropriate answers to the waste questions and to avoid water quality degradation, a thorough and critical assessment of a technology's impacts before it is adopted is the only reasonable approach to economic development if we are to preserve life as we know it and if we are to avoid burdening our future generations with the overwhelming costs of massive remediation. In a famous American advertisement the consumer is warned, "You can pay me now or you can pay me later." As we have seen in the North American Great Lakes region, paying later means accepting economic and social costs far greater than might have been necessary if better informed advance planning had taken place. We have sufficient experience with technology so that persons with relevant expertise can now

---

* L. Winner, AUTONOMOUS TECHNOLOGY 229, 238 (1977).

** G. Hardin, FILTERS AGAINST FOLLY: HOW TO SURVIVE DESPITE ECONOMISTS, ECOLOGISTS, AND THE MERELY ELOQUENT 67-68 (1985).

engage in relatively sophisticated analyses of the likely impacts in a number of affected sectors. To attain the goal of sustainable economic development, decision-making should be guided by and founded on comprehensive environmental evaluations.

As environmental impact assessment strategies are developed, fixing goals for the assessment process is a critical first step. From society's perspective, a particular technology may be desirable if it increases the quality of life more than it diminishes it. Recognizing that this perspective is not always shared by the private sector where private gain may be assigned the highest priority, this discussion adopts the view ascribed to society in general. In the area of environmental protection, inherent in the notion of the enhancement of the quality of life is its preservation through adequate environmental protection measures.

Reviewing the environmental impacts of technology as we have thus far been able to discern them in the North American Great Lakes, it is clear that the choice of appropriate technology to spur economic development should move in the direction of elimination of the waste stream, and it should be carried out in ways which conserve scarce natural resources, conserve scarce financial resources, and preserve human health. Planners around the world can learn from the mistakes made in these lakes, taking steps to preserve water quality, steps that will significantly reduce total development expenditures.

Owing to the critical nature of the environmental assessment process, all policymakers with responsibilities encompassing or touching on environmental issues ought to take every available opportunity to build assessments into decision-making processes, in the public and private sectors, from the broadest authorities to those most narrowly circumscribed. The first all-media, U.S. federal environmental legislation adopted in 1969, the National Environmental Policy Act* (NEPA), requires that impacts be considered, but revelations of the assessment process need not affect project planning or implementation. In this frequently cited but timid approach, the United States requires only a "detailed statement"

---

* National Environmental Policy Act, 42 USC 4321-4391 (1983).

(now called "Environmental Impact Statement") on "recommendations or reports on proposals for legislation or other major federal actions significantly affecting quality of the human environment."*

In order to widen the sweep of such requirements, smaller governmental units with the requisite authority, such as states, provinces, cantons, cities, counties, villages, and townships, should require environmental assessments for projects regulated by them. To encourage assessments in the private sector, zoning ordinances, economic development grants, tax holiday and abatement processes could all contain assessment requirements. Tax codes could be written to reward environmentally conscious entrepreneurs and institutions. Moreover, through government regulation of financial institutions, environmental assessments could be required as a condition of loans for private development projects. The extra-territorial impact of development must also be addressed. Increasingly treaties include environmental requirements. Generally, treaty provisions are narrowly tailored and they are not universally incorporated. Such provisions ought to be broadened and become universally adopted. The foregoing enumeration provides examples of the available avenues environmentally-conscious decision-making.

Environmental assessments should also be required for internationally funded development projects, so important in the Third World where in most countries economic development is in its early stages and the environment is currently under attack on many fronts.** A number of widely circulated reports and litigation have brought to public attention criticism of the World Bank and U.S. Agency for International Development funded projects,*** spurring formal adoption of assessment processes in these

---

* National Environmental Policy Act, 42 USC 4332 (C) (1983).

** See e.g., Lloyd Timberlake, AFRICA IN CRISIS: THE CAUSES, THE CURES OF ENVIRONMENTAL BANKRUPTCY (1985).

*** See,e.g., Int'l. Inst. for Env. and Development, Banking on the Biosphere (1978); Sierra Club, Bankrolling Disasters: International Development Banks and the Global Environment (1986). The U.S. Agency for International Development was sued for its failure to take into account the environmental impacts of its development activities. NRDC v. U.S. Agency for International Development, 6 Env. L. Rptr. 20121.

agencies.* While it may take time for entrenched bureaucracies to respond in an effective fashion to these new environmental requirements, the fact of the agencies' reorientation is cause for cautious optimism.

The successful use of environmental assessment in this context can serve as a model and source of encouragement for countries receiving aid from development assistance agencies to adopt similar requirements. Learning the value of such assessments in the development process, local governments will be much more apt to follow suit when they have observed the benefits at close hand and can see the positive results flowing from assessments. The United Nations and member nations should strongly encourage all multilateral and bilateral lenders to follow suit when providing funding for economic development.** United Nations agencies whose missions may affect the environment also have important roles to play providing information, expertise, and start-up funds.

In a thorough discussion of the value of and need for environmental impact analysis, attention must be devoted to the configuration of the analysis itself. To be of genuine assistance in decision-making, environmental assessments must be carefully and thoughtfully conducted by teams of independent experts. Persons with no personal stake in the outcome are more likely to provide a useful and objective evaluation and thus, should be chosen.

The use of advisory bodies can provide an additional safeguard to protect assessment independence. With representatives from the various stakeholder groups, pressure for a biased result is less likely to be used. In such an atmosphere, technical professionals will be in the best position to provide their objective, expert opinions. The Office of

---

* As a result of a settlement of NRDC's lawsuit, the agency promulgated environmental procedures rules.

In 1988, the World Bank increased its environmental staff from six persons to 60.

** Some bilateral lenders already maintain some type of environmental assessment component to their lending procedures. Food and Agriculture Organization, ENVIRONMENTAL IMPACT ASSESSMENT AND AGRICULTURAL DEVELOPMENT 55-56 & 61 (1982).

Technology Assessment of the United States Congress (OTA) routinely uses this approach and finds that it enhances evaluation reliability.*

In order to increase further the level of independence of these teams and to lower the costs of creating and staffing them, particularly in less developed areas where human and financial resources are severely limited, multinational teams should be used. With experts from a number of countries, assessment teams would be less likely to bow to local pressure. In addition, especially in areas where countries share a number of physical or social characteristics, efficiency and economy would seem to dictate such an approach. As local expertise is built to facilitate appraisals where none could occur due to lack of human and financial resources, multinational groups would learn from each other, enhancing international understanding.

As well as being independent, given the wide variety of potential environmental impacts of economic development projects, assessment teams should be staffed by representatives of many disciplines to make comprehensive evaluations. For example, in evaluating a proposed lakeside manufacturing project, hydrologists, aquatic biologists, geologists, social scientists, air and water chemists, economists, and plant and wildlife ecologists would provide factual information and perspectives essential to a sufficiently thorough environmental assessment. The OTA has provided a number of well received assessments based on this type of comprehensive, multidisciplinary approach. A comparatively small agency,** OTA relies primarily on the data gathered by outside consultants with whom it contracts and who typically number 200 persons.***

A third key ingredient to an effective environmental assessment process is extensive interaction and consultation with persons who will be affected by the project. One could argue that principles of justice and fairness dictate a cooperative planning process which incorporates local

---

* Gibbons (the OTA's director) & Gwin, Technology and Governance, 7 Technology in Society 341 (1985).

** Currently the OTA staff has fewer than 150 permanent employees, including support staff.

*** Information provided in an interview on July 14, 1988, of OTA's Drs. Alison Hess, Project Director/Analyst, and Walter Param, Program Manager, Food and Renewable Resources Program.

input. It is important to recognize in addition, however, that local people possess information essential to complete and reliable assessments. The development literature is replete with numerous examples of projects that failed because local people were either not consulted or their advice was ignored.* A solid understanding of local culture and physical resources that can only be obtained from local sources is crucial to development success. Failure to bring affected persons into the decision-making process at a minimum stimulates a negative response to the project and more likely results in a poorly conceived project.

As obvious as the value of the environmental assessment process may be, it nonetheless has its detractors. A frequent criticism of and obstacle to environmental assessment is its cost. Perhaps because it is a relatively new concept and thus is viewed as an add-on expenditure, critics focus on the difficulty of seeking additional funding or having to reduce existing limited funding for other aspects of the project. Given its critical nature, however, it should be viewed as the sine qua non of all projects and should be the first line item in the pre-planning budget.

Precedents for such funding priorities are abundant. For instance, U.S. inheritance statutes typically contain provisions requiring that funeral expenses hold a superior right to estate funds. U.S. secured transaction statutes establish payment priorities to protect lenders. In both of these examples, priorities are established to create incentives for socially desirable behavior. Similarly, if behavior which recognizes the high priority of environmental values is to be fostered, funds for environmental impact evaluations must be given legal priority, consonant with recently adopted environmental protection policies around the world. In sum, the expenses of environmental assessment should be viewed as an

---

* WOMEN CREATING WEALTH: TRANSFORMING ECONOMIC DEVELOPMENT, Rita S. Gallin & Anita Spring, eds. (1984); Ross, Collaborative Research for More Effective Foreign Assistance, 16 World Development, No. 2, at 231-36 (1988).

essential, integral part of any development activity.* Furthermore, as investigation of environmental questions becomes integrated into project planning, its cost will be lower by virtue of its anticipation by all those involved in the project. The need for redoing any phases of the project, including planning, will be minimized.

Environmental assessment requirements have also been criticized for degenerating into a useless and time-consuming paper exercise. The danger, from the perspective of those seeking to protect the environment and those seeking economic efficiency, is this: the goal of environmental preservation is supplanted by the goal of producing a satisfactory and acceptable document so that the project may proceed as planned. Rather than focusing on merely describing the impacts identified and the risk factors associated with them to satisfy the assessment requirement, strategies must be developed to force project planners to go beyond identification and description to mitigation and avoidance.

From the perspective of environmentalists, technical environmental assessments may conceal another problem. Technical experts can manipulate methodologies and data in ways that disguise or conceal subjective underlying assumptions. Thus, actual impacts may be understated to permit development to go forward.** In the final analysis, both developers and environmentalists must recognize that assessments, even the most sophisticated, are predictions. Consequently, to achieve the highest possible degree of assessment reliability, decision-makers must strive for comprehensive and thorough assessments growing out of malleable processes designed to permit adjustment to particular projects.

Having pondered the value of the assessment process and approaches to strengthening it, we must consider methods to bring about its adoption and

---

* Given the mismatch in the huge number of projects world-wide and scarce financial resources, in the short-run at least a threshold needs to be established which once achieved guarantees environmental assessment scrutiny. This issue can probably best be addressed through the formulation of a series of thresholds based on such factors as project cost, project size, and sensitivity of the area.

** Wynne, At the Limits of Assessment, in TECHNOLOGY ASSESSMENT AND QUALITY OF LIFE, G.J. Stober and D. Schumacher, eds., 284 (1973).

use. A broad category of viable strategies is legal and economic incentives to environmentally responsible behavior, both in planning and project implementation. To apply effective legal and economic pressure, statutes should include provisions forcing project modification or cancellation when appropriate to prevent environmental damage. To achieve the goal of sustainable development and environmental preservation, assessment requirements should include provisions that permit orders by appropriate courts or agencies to modify or abandon projects based on evaluation results. Such potential outcomes will offer strong incentives to planners and developers to investigate and weigh seriously environmental impacts and to plan in the least environmentally damaging way possible to preserve the project.

Traditionally in environmental legal battles, those defending the environment have to bear the burden of proving the proposed action unacceptably harmful, forcing plaintiffs who sue to protect the environment into an uphill struggle. Under current law, because of the burden of proof rule, in close cases, environmental protection claims fall before the developer's bulldozer. In cases where environmental values are at stake, this rule should be reversed. Based on current scientific knowledge about the widespread harm that may ensue from damage to ecosystems, economic development decisions ought to err on the side of preserving the environment. Those advocating particular development projects should be required to prove by substantial evidence that they are not damaging or that mitigation strategies can be effectively employed to minimize damage to an acceptable level.

The imposition of statutory strict liability for harm caused acts as a powerful financial incentive to environmentally safe development. It is based on a simple and yet fundamental legal principle: those who cause harm must bear responsibility for their actions or, more succinctly and specifically, polluters pay. Under strict liability theory, the plaintiff need only establish that the defendant caused the injury. The more difficult legal argument to win, proving defendant's fault or negligence, is removed from the case, making plaintiff's compensation, and environmental remediation, easier to achieve.

In situations where injury has the potential to be great and widespread, the shield of organizational form should be removed. Statutory provisions creating personal liability for corporate employees, officers, and directors who can be charged with knowledge of and control over the harmful activity are also crucial to inducing right action. A world-wide movement in this direction is slowly gaining ground.* For example, more and more U.S. environmental statutes contain criminal provisions that are being enforced.** Recent Nigerian action which exacts the death penalty for importation of toxic waste*** also demonstrates a growing global consensus that strong steps must be taken to avoid further damage.

Imposition of liability alone, however, is insufficient to prevent pollution or to secure payment for injuries. Substantial financial deposits should be required of private foreign concerns seeking to locate facilities. Presumably, with substantial amounts at stake and in the control of the host country, firms would have a greater incentive to act responsibly. In the event that pollution injuries nonetheless occur, a fund to cover some of the losses would be readily accessible. One model for such deposits exists in U.S. states' regulation of insurance companies. To protect in-state policyholders, out-of-state insurers are required to deposit substantial amounts accessible to the insurance regulatory agency in states in which the insurers are not incorporated.****

In addition to legal sanctions, policymakers in regions seeking economic development need to identify and implement alternative positive inducements to locate in their area. Unfortunately no magic formula exists to which today's less developed countries can look to attract the sustainable economic development they so desperately need. Creating a conducive business climate in the Third World without courting disaster will take much creativity and effort.

---

* In response to one of the largest environmental disasters ever, in December, 1984 in Bhopal, India, the company's chairman and two senior officials were arrested and company records were seized. S. Hazarika, India Police Seize Factory Records of Union Carbide, N.Y. Times, Dec. 7, 1984, at A1, col. 3 & A10, col. 1; Reinhold, Indians Arrest and Then Free U.S. Executive, N.Y. Times, Dec. 8, 1984, at A1, col. 2 & A7, col. 3.

** The owners of a mushroom farm were convicted, fined, and sentenced to short jail terms for violating the U.S. Clean Water Act. U.S. v. Frezzo Bros., Inc., 602 F.2d 1123 (1979), cert. denied 444 U.S. 1074 (1980).

*** Brooke, Waste Dumpers Turning to West Africa, N.Y. Times, July 17, 1988, at 1 & 7, cols. 1 & 1.

**** E.g., Mich. Comp. Laws Ann. 500.411 (West 1983).

Providing a skilled work force offers yet another way of attracting development. The ready availability of dependable, skilled workers is an important factor in labor-intensive operations location decisions. Thus, investment in education can lead to job creation. Educated workers are self-supporting, pay taxes, and are less likely to harm the environment. Through education they recognize environmental values, and through employment, they are not engaged in a life and death struggle to survive from day to day which may necessitate destruction of their habitat.

Having determined that environmental preservation through sustainable economic development is a global necessity, the political obstacles to this objective must be acknowledged and strategies to overcome them must be devised and pursued. Such obstacles include rivalries between factions within a single country, competition between countries, and corrupt bureaucrats and politicians. This may be the single most intractable issue in the Third World. If sustainable development cannot be achieved through the leadership of those currently in power, it must be accomplished in spite of their greed, incompetence, and ignorance. Education again comes to the fore. If people understand what is at stake, they will pressure leadership. In making a realistic assessment of the degree to which citizens can participate in formulating policy, however, the dire straits of a large percentage of persons in the Third World must be taken into account. In order for education to build awareness and understanding, a decent standard of living must be assured for all people. Only those whose basic needs are met have energy to devote to environmental concerns. Thus, world-wide environmental protection planning must include steps to address impediments to achieving this goal. These steps must be addressed to reducing foreign debt, improving health care, creating employment opportunities, and reducing population growth.

Countries acting alone in adopting any or all of the measures proposed above risk loss of potential investment and development. Thus, paths to concerted action must be explored and pursued. Regional, continental, and global political organizations must commit themselves to assisting member governments to make environmental preservation a real priority rather than merely a part of their political rhetoric. Political unity, spurred by essential environmental education of all citizens, is essential to protect against intolerable economic losses in some countries and the creation of pollution havens in others.

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

OPTIMUM UTILIZATION OF LAKES

A. DIÓSI

Ministry for Environment and Water Management
H-1011 Budapest, Fő utca 44-50, Hungary

IMPORTANCE OF LAKES AND THEIR MAIN FUNCTIONS

In this presentation the term "lake" is used to denote all natural and man-made inland bodies of standing water from ponds to the Caspian Sea.

The value of these lakes depends on size, water quality and natural setting.

Lakes can help satisfy some of our basic needs (e.g. drinking water and food), and many other human requirements; they can also have a high scientific value, as well as a great cultural and emotional, sometimes even spiritual importance. In fact, lakes make a unique natural resource, which can hardly be substituted by other kinds of resources.

It is important to note that the land surface occupied by lakes is quite significant. In Japan 1.65 per cent of the country is covered by lakes, in Britain and in Hungary, as well, lakes occupy around 1 per cent of the area, but the same figure for Nicaragua is as high as 7%, for Finland 9.4%, and for Malawi 20%.

All lakes have various functions: they have to deal with different kinds of anthropogenic effects, which they are exposed to, and are also subject to many types of practical uses by man. Their uses vary in each case, but can include some or many of the functions specified in Table 1.

Table 1. Functions of lakes

As a site for recreation	As a water resource	As a biological resource	As a nature conservation area
Swimming	Water supply	Fish breeding in the open lake	Conservation of
Sailing	- potable water		- aquatic plants
Boardsailing	- industrial water	Caged fish farming	- aquatic animals
Rowing	- irrigation water	Mussel farming	Conservation of birds
Water skiing	Elimination of pollutants of	Poultry keeping	Conservation of scenery
Angling		Reed and other plant production	
Walking by the side	- direct discharges		
	- sewage effluents		
Bird-watching	- air-borne substances		
Therapeutic use	Industrial cooling		
	Flood storage		
Ice-skating	Hydro-power generation		
Ice-sailing			
	Water transport		
	Raw material supply (e.g. soda ash)		

ANTHROPOGENIC EFFECTS ON LAKES

Anthropogenic effects, which are due to the various functions of lakes, or are rather the results of man's different uses of them, can be divided into five groups:

I. Direct impacts inside the lakes.

II. Effects of catchment-based human activities.

III. Effects of water transfer to the lake's catchment from another catchment area.

IV. Water pollution due to air-borne pollutants.

V. Effects of management and control measures applied to lakes.

Direct impacts inside the lakes

1. Pollution made inside the lakes, as follows:

- Organic load due to different kinds of fishing, e.g. commercial fishing, fish farming in fish cages, angling (if anglers feed fish at favoured places).
- Oil slicks coming from motor boats.
- Accidental spillage of chemicals.
- Illegal spillage of chemicals.
- Polluting effects of too many bathers wading in the soft bottom of the lake, who thus stir up mud.
- Even suntan oil can make a significant water pollution in some cases.
- The most important of all pollution types occurring inside a lake, however, is the so-called "internal load", which normally means the stored nutrient content of a lake, available for further biological processes, but can also be the waste material (hazardous wastes, etc.) content settled on the bottom of the lake.

2. Infection of the lakes by certain worms causing typical water-based diseases

- Schistosomiasis (bilharziasis)

This is a disease widespread in Africa, but it also occurs in South America and countries in the Far East.

Man can be infected either by drinking infected water or by standing or bathing in such water.

- Guinea worm disease

  This disease is common in Africa and India.

  Man can be infected by swallowing infected water fleas (the intermediate host) when drinking water.

- Tropical liver fluke disease

  This disease is common in countries in the Far East.

  Man can contact the disease by eating under-cooked fish infected by the cyst of the fluke.

3. Introducting alien animal or plant species to the aquatic environment of lakes

- Introducing foreign species of aquatic animals, particularly fish, can cause a lot of disturbances in the original ecosystem of lakes.

- Foreign aquatic plant species when introduced to an environment, which is a new but otherwise suitable habitat for them, can also cause great troubles for the lakes. The problem of the proliferation of water hyacinth (Eichhornia crassipes) is a good example of that.

Effect of catchment-based human activities

1. Point source water pollution in the catchment area derived from municipal and industrial sewage, and from liquid manure of feedlots.

2. Non-point source water pollution in the catchment area of both agricultural and municipal origin.

3. Polluting effects of inflowing streams which are receiving water pumped from mines or discharged by land drainage. (Though, sometimes the extracted mine water can improve the quality of the recipient.)

4. Increase in the concentration of salts and other harmful substances of the lakes due to excessive abstractions from the rivers feeding them. The plight of Lake Aral is an example of that.

Effects of water transfer to the lake's catchment from another catchment area

1. The quality of water received from another catchment area can cause difficult problems in a lake, although it can be an advantage in this respect also.

2. Even the unusually high flow of the introduced "foreign" water itself can cause problems in the lake's catchment, e.g. erosion, change of the original ecosystem, etc.

Water pollution due to air-borne pollutants

1. Chemical (acid, nutrient, etc.) deposition in the lakes, due to local air pollution downwind from the sources of pollution.

2. Chemical (acid) precipitation derived from long-range transboundary air pollution.

3. Radioactive atmospheric fall-out.

4. Global climatic changes due to air pollution, which might alter the temperature of lakes or otherwise affect their plants and animals.

Effects of management and control measures applied to lakes

To get the highest possible benefit from an intensively used lake, one has to alter some of its features or otherwise interfere with it, thus making further harmful effects for the lake to deal with. The management and control measures normally used for this purpose can be divided into three groups: alteration of the main features of lakes, water level adjustment, and introduction of chemical compounds.

1. Alteration of the main features of lakes

- Realignment and transformation of the shorelines by making artificial or semi-artificial shore defence works and/or constructing harbours, water intakes, etc. Although these constructions have a useful function, their disadvantages are many: they adversely affect the biology of the lake, reduce the self-cleansing function of the same, and even decrease the natural beauty of the area.

- Dredging of a lake is normally required to remove bottom sediment originating from man's polluting actions, or to stop or turn back the natural ageing of the lake. These actions, although helpful in one respect, create a problem of adjustment for the lake, which normally cannot adapt to sudden changes.

2. Water level adjustment

Some of the impounding reservoirs and even lakes are subject to short-term regulation of the flow released from or arriving at them, a management technique which is useful for peaking power generation but definitely badly affects the reservoir or lake as an ecosystem.

On the other hand, if a lake has for thousands of years been exposed to yearly fluctuations of water level, it can certainly be harmed by the adjustment of the incoming flow, which stabilizes its water level. The harm again is done to the original flora and fauna of the lake, while the expected advantage is a fixed and more easily accessible shore for the swimmers.

3. Introduction of chemical compounds

- Liming is used to reduce the extremely high acidity of some lakes.
- Nitrogen addition. In case of nitrogen limitation the addition of nitrogen may help stop the proliferation of nitrogen fixing blue-green algae. However, great care must be taken with the timing and dosage of this nutrient.
- Application of cetyl alcohol. Some chemicals applied on the surface of a reservoir can cover it and reduce evaporation losses. Cetyl alcohol (1-hexadecanol) is such a chemical.

<u>Environmental impact assessment</u>

To estimate the possible environmental impacts of the suggested management and control measures on existing lakes, and particularly to assess the environmental effects of a proposed man-made lake, or the draining of an existing one, would require a systematic approach. This can best be done by applying to the case the method of Environmental Impact Assessment, a very useful tool that I am not going to detail in this presentation.

THE EFFECTS ON LAKES OF TOXIC CHEMICAL COMPOUNDS OF NATURAL ORIGIN

In some cases toxic chemical compounds of natural origin appearing irregularly or completely unexpectedly in a lake or its catchment area can cause trouble, even a havoc for that lake.

These "natural pollution" forms can mostly occur in volcanic regions, either as dissolved salts, or dissolved gases. The gas burst of Lake Nyos in Cameroon, in 1986, in which 1700 people and more than 3000 cattle died, is a sad example of dissolved gas (carbon dioxide) accumulation and subsequent escape from a lake.

LAKE UTILIZATION BY AN INTEGRATED MANAGEMENT SYSTEM

Ownership of lakes

Lakes are, or should be, assets not liabilities of a country. Yet, in many instances, instead of bringing in revenue, they make a burden on a nation's budget, because they are not utilized well enough, or are just mismanaged.

The main problem of lakes, in my opinion, is the lack of owner's mentality in their management. In fact, they rarely have a rightful owner, who has got full control over them, and takes care of all aspects of their use, management and control.

Ownership is well established on a national level: lakes either belong to one nation, or are shared by two or more - they are not international waters. However, on the management level - unlike agricultural land, where there is always an owner - lakes are a prey to all comers.

Lake utilization options

First of all, I want to categorically state that the functions of lakes mentioned in Table 1, in actual fact are services. Therefore, like all services, the functions of lakes should also not be provided free of charge, but for remuneration.

For example, all polluters who discharge waste water into a lake should pay for the treatment made by the lake, even though that treatment can be called "self-purification".

The functions of a lake basically depend on water quality; I mean the water quality is the decisive factor in a lake's use. Thus, one can distinguish

- sewage lagoons/waste disposal lakes,
- fish ponds/fish breeding lakes, and
- bathing water quality lakes.

When the water quality of a sewage lagoon or a grossly polluted lake has improved, it can be made a fish pond; if a lake (or fish pond), which has been used for intensive fish breeding has got its water quality greatly improved, it can be turned to a bathing lake. The reverse process can also happen, in fact it does happen, when the water quality is deteriorating.

In large lakes, however, there can be different water quality areas in the same lake. Normally these are basins, or parts of the coastal water area.

In this context, I want to point out that water quality deterioration can be due not only to water pollution and nutrient induced eutrophication, but to disease-causing organisms too, i.e. pathogenic microorganisms, and worms bringing about the water-based diseases mentioned earlier.

Thus, lakes and reservoirs which were ideal places for tourism development could only be exploited for fishing and other inferior water quality uses in case of schistosoma and guinea worm infection. However, if the liver fluke is present in a lake, not even fishing can be a viable option.

Proposed lake management

As for the current lake management practice, I would call it "management by crisis". If many things have definitely gone wrong, only then is action taken to "save the lake", that is to put it back to its original functions.

Instead of this wrong and greatly inefficient practice, a systematic approach is proposed in which lakes are considered multi-functional units. In this system, based on water quality (the decisive factor), a suitable combination of the various functions is established.

To this effect one should work out a list of all major existing functions, as well as all proposed uses of a lake. Then, to find out the interrelationships amongst the functions of the lake, a diagrammatic representation is suggested. Based on these particulars, one can decide about the priorities to be pursued in a particular case by trying to reconcile the conflicting interests arising there.

Examples to illustrate the above ideas for Lake Balaton and Lake Kis-Balaton are presented in Tables 2 to 5 and Fig. 1.

CLASSIFICATION OF LAKES AND THE DRAWING UP OF A WORLD LAKES CATALOGUE

The purpose of establishing a World Lakes Classification System and Catalogue is to help the managers (owners) of lakes find all data, which are available on other lakes, so that they can learn from the experience of others who are, or have been faced with similar problems.

Classification of lakes

Classification of lakes can first of all be made on size, starting with bigger-sized sewage lagoons and fish ponds and ending up with the Caspian Sea, as suggested in Table 6.

- Class I is the category of very small lakes, the surface area of which is between 0.5 and 100 hectares. Apart from small-sized natural lakes, many man-made lakes belong to this category, such as smaller reservoirs, gravel pit lakes, lakes of other open mine excavations, by-passed meanders, upper reservoirs of pumped storage plants, fish ponds, and sewage lagoons, etc.

I wish to mention in this context that by-passed meanders are quite significant for Hungary; all in all they cover 114 km^2, which is more than one-sixth of the surface area of Lake Balaton.

Table 2. Major functions of Lake Balaton

RECREATION
Swimming
Sailing/shipping (passengers)
Walking/open air activities
Ice-skating/ice-sailing
WATER RESOURCES MANAGEMENT
Water supply
Receiving treated sewage effluents
Receiving wastes of direct discharges
Shipping (cargo)
BIOLOGICAL RESOURCES MANAGEMENT
Angling
Commercial fishing
NATURE CONSERVATION
Landscape preservation
Protection of reedbeds
Conservation of species

Table 3. Major functions of Lake Kis-Balaton

RECREATION
Bird-watching
WATER RESOURCES MANAGEMENT
Receiving (treating) sewage effluents
Water storage
BIOLOGICAL RESOURCES MANAGEMENT
Commercial fishing
Reed production
NATURE CONSERVATION
Protection of birds
Protection of other species

Table 4. Priorities of functions and uses of Lake Balaton

Priority groups	Functions and uses in order of importance
Main functions	Swimming
	Sailing/shipping (passengers)
	Walking/open air activities
	Protection of reedbeds
	Water supply
	Landscape preservation
Secondary functions	Receiving treated sewage effluents
	Receiving wastes of direct discharges
	Angling
Unimportant functions*	Commercial fishing
	Shipping (cargo)
	Conservation of species
	Ice-skating/ice-sailing

Table 5. Priorities of functions and uses of Lake Kis-Balaton

Priority groups	Functions and uses in order of importance
Main functions	Receiving (treating) sewage effluents
	Water storage
	Protection of birds
Secondary functions	Protection of other species
	Commercial fishing
Unimportant functions*	Reed production
	Bird-watching

*Note: "Unimportant functions" are unimportant but this time. Every new situation presents new opportunities, and makes new challenges to selecting our priorities.

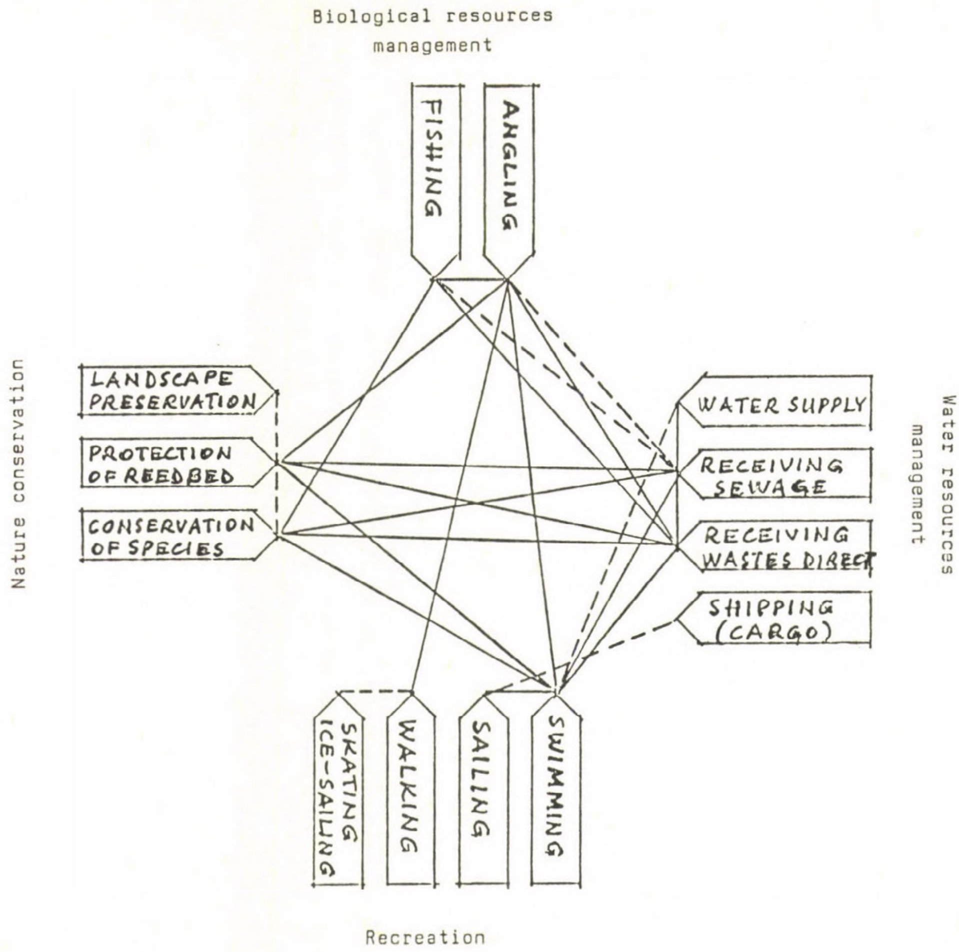


Fig. 1. Interrelationships of various functions
and uses of Lake Balaton

Note: Full lines represent conflicting interests and
dotted lines non-conflicting interests.

Table 6. Main classes of lakes

Class	Designation	Surface area (ha)	(km^2)
I	Pond	0.5-100	
II	Smaller lake or reservoir	100-10,000	1-100
III	Medium-sized lake or reservoir		100-10,000
IV	Great lake		10,000-100,000
V	Sea		above 100,000

- Class II is the category of smaller lakes and reservoirs. Most of the world's natural lakes belong to this group of lakes, and so do many of the man-made ones also.

Contrary to Class I lakes, the water quality of which is uniform throughout, lakes belonging to this category may have different water qualities inside the lakes and along the shores, particularly if they are intensely used for bathing.

- Class III lakes can be called medium-sized lakes, because they are neither small, nor very great (compared to the great lakes), yet, some of them are relatively large, e.g. Lake Titicaca (8,135 km^2), or Lake Turkana (6,400-8,000 km^2).

Many of the well-researched lakes belong to this group, including Lake Balaton and Lake Biwa.

- Class IV lakes are the real great lakes, including the Laurentian (Great) Lakes in North America, the East African great lakes and a few others.

The total number of these great lakes is 18 together with Lake Onega (9,943 km^2). Lake Maracaibo (12,950 km^2) would also belong to this category, bringing the total number of these lakes to 19, however, this water body is normally not considered a real lake, but a bay of the Gulf of Venezuela.

- Class V is a unique case, because there is only one lake (in fact closed inland body of standing water) in this category; this is the Caspian Sea, with a surface area of 436,400 km^2.

Further classification depends upon other characteristics. Thus one can distinguish natural or man-made, deep or shallow, cold climate or hot climate lakes.

Distinction between cold climate and hot climate lakes is based on the fact that deep cold climate lakes are having autumn and spring turnovers, while the hot climate lakes do not, but, apart from that, all cold climate lakes have a winter period during which their behaviour is much different than that in the other seasons.

Practical considerations of the establishment of the World Lakes Classification System

The World Lakes Classification System cannot be set up in itself, without the active support of all member nations interested in this type of co-operation, rather it should solidly be based on national data.

However, to get the maximum benefit of this system, national data must be made up of figures comparable with those of other nations.

This way, lakes - the last inefficiently used important natural resource - can be much better utilized for the benefit of all owner countries.

MODELLING IN LAKE MANAGEMENT

Symp.Biol.Hung. 38 (1989)
Akadémiai Kiadó, Budapest

Conservation and Management of Lakes
J. Salánki, S. Herodek (eds)

EUTROPHICATION MODELS: THE STATE-OF-THE-ART

S.E. JØRGENSEN

Department of Pharmaceutical Chemistry AD,
Royal Danish School of Chemistry
DK-2100 Copenhagen, Denmark

1. INTRODUCTION

Eutrophication of lakes and BOD/DO-problems of rivers are the two most modelled environmental problems. A number of models have been developed and applied and also a large number of case studies.

This paper attempts to answer the question: where are we today in the use of eutrophication models as a tool in environmental management and in ecological research of lakes?

2. BRIEF ABOUT THE HISTORY OF EUTROPHICATION MODELS

Jørgensen (1983) reviews the mechanistic models developed during the seventies, and Straskraba and Gnauck (1985) present an overview of eutrophication models. They have updated the summary published by Chahuneau et al. (1980). It is reproduced with permission in Table 1.

These references give a good introduction to the history of eutrophication models, and here it will be attempted to summarize the experience gained through the many eutrophication model studies.

It has been found of great importance to follow a modelling procedure, which at least considers (see also Jørgensen 1986)

(1) the problem and the ecosystem characteristics,

(2) a conceptualization of the model,

(3) parameter estimations based upon literature values and parameter calibration, and

Table 1

Eutrophication models; according to Chahuneau et al. (1980), updated. Papers referred by Chahuneau et al. marked with * are not given in references unless otherwise cited. Biological structure denoted by letters B, C and S; hydrodynamic structure characterized by Roman numerals. B - biologically simple models, no breakdown within trophic levels; C - biologically complex models including several species at various trophic levels; S - selforganisation models; I - hydrodynamically undifferentiated models (only one homogeneous layer); II - one-dimensional models with vertical gradients; III - two- and three-dimensional models with vertical and/or horizontal gradients; TC - constant; TV - variable nutrient ratio in phytoplankton

Author	Model structure	No. of state var.	Nutrients	TC, TV	Sediment effect	Oxygen	Number of species				
							Phytoplankton	Zoopl.	Fish	Benthos & macrophyt.	Destruents
Brezonik (1968)	B I	7	N	TC	+	-	1	1	1	1	1
Parker (1968)	B I	4	P	TC	-	-	1	1	1	0	0
Larsen et al. (1974)	B I	3	P, N	TC	+	-	1	0	0	0	0
Jørgensen (1976); Model 1	B I	9	P, N, C	TC	+	-	1	1	0	0	0
Jørgensen (1976); Model 2	B I	12 (2)	P, N, C	TV	+	-	1	1	0	0	0
Nyholm (1977)	B I	7	P, N	TV	+	-	1	0	0	0	0
Watanabe (1978)	B I	11	P, N	TC	-	-	1	1	0	0	0
Gentil (1979)	B I	7	P, N	TC	-	-	1	1	0	0	0
Verhagen (1979)	B I	8	P, Si	TC	-	-	3	1	0	0	0
Vavilin (1979)	B I	2	P	TC	-	-	1	0	0	0	0
Blake & Gentil (1979)	B I	7	P, N	TC	-	-	1	1	0	0	0
Vavilin (1980)	B I	4	P	TC	+	-	1	0	0	0	0
Krambeck & Krambeck (1980)	B I	4	P	TC	-	-	2	1	0	0	0
Straskraba et al. (1980) AQUAMOD 1	B I	3	P	TC	-	-	1	1	0	0	0

Table 1 (continued)

Author	Model structure	No. of state var.	Nutrients	TC, TV	Sediment effect	Oxygen	Number of species: Phytoplankton	Zoopl.	Fish	Benthos & macrophyt.	Destruents
Menshutkin (1973)	B II, 2	11	without spec.	TC	+	+	1	1	1	0	1
Imboden (1973)	B II	3	P	TC	+	-	1	0	0	0	0
Gargas (1976)	B II, 3	8	P, N	TC	+	+	1	1	1	0	0
Bradstetter et al. (1977)	B II, n	12	P, N	TC	+	+	1	1	0	0	0
Straskraba & Dvorakova (1977)	B II	4	P	TV	+	-	1	1	0	0	0
Guliver & Stefan (1982)	B II, n	2	P	TC	-	-	1	0	0	0	0
Straskraba (1979 a) AQUAMOD 2 (Chap. 11.)	B II, 2	5	P	TC	+	-	1 + 1	1	0	0	0
Benndorf & Recknagel (1979) SALMO	B II	3	P	TC	-	-	1	1	0	0	0
Straskraba (1980 b) AQUAMOD 3	B II, 3	8	P	TC	+	-	1 + 1	1	0	0	0
Di Toro et al. (1971)	B III	7	P, N	TC	-	-	1	1	0	0	0
Najarian & Harleman (1975)	B III (n)	7	N	TC	-	-	1	1	0	0	0
Ikeda & Adachi (1978)	B III (5)	6	P, N	TC	-	-	1	1	0	0	0
Knoblauch (1978)	B III	12	P	TC	-	-	2	1	0	0	1
Leonov (1980)	B III	5	P	TC	+	-	1	0	0	0	1
Garcon (1981)	B III (n)	10	P	TC	+	-	3	1	0	0	0
Kutas & Herodek (1982) BEM	B III	9	P, N	TV	+	-	3	0	0	0	1

Table 1 (continued)

Author	Model structure	No. of state var.	Nutrients	TC, TV	Sedi- ment effect	Oxygen	Number of species: Phyto- plankton	Zoopl.	Fish	Benthos & macro- phyt.	Des- tru- ents
Parker (1974)	C I	1	P, N	TC	-	-	2	2	0	0	0
Park et al. (1974)	C I	28	P, N	TC	+	-	2	3	3	6	1
Lehman et al. (1975)	C I	18	P, N, Si	TV	-	-	5	0	0	0	0
Patten et al. (1975)	C I	33	P, N, C	TC	-	+	7	2	9	4 + 12	0
Gupta & Houdeshell (1976)	C I	13	P, N, Si	TC	-	-	3	2	2	0	1
Bierman (1976)	C I	20	P, N, Si	TV	-	-	4	2	0	0	0
Vietinghoff et al. (1979)	C I	12	N, Si	TV	-	-	3	2	0	2	1
Los (1980) BLOOM II	C I	14	P, N	TC	-	-	10	1	0	0	0
Schellenberger et al. (1983)	C I	15	P, N	TC	+	-	3	1	1	0	1
Chen & Orlob (1975)	C II	18	P, N, C	TC	-	-	2	1	3	1	0
Thomann et al. (1975) Lake 1	C II	10	P, N	TC	-	-	1	2	2	0	0
Thomann et al. (1975) Lake 2	C II	15	P, N, C	TC	+	-	1	2	2	0	0
Park et al. (1975) CLEANER	C II	14-40	P, N (C, Si)	TC	+	-	3	3	3	6	1
Canale et al. (1976)	C II	25	P, N, Si	TC	-	-	4	9	0	0	0
Scavia et al. (1976)	C II (n)	17	P, N, C	TC	-	-	4	6	0	1	0
Park et al. (1979)	C II (2)	31	P, N	TC	+	+	3	3	3	2	0
Chen et al. (1975)	C III	33	P, N, C	TC	-	-	3	4	4	2	0
Thomann et al. (1975) Lake 3	C III	15	P, N	TC	+	-	1	2	2	0	0
Schellenberger et al. (1978)	C III	17	P, N (Si)	TC	+	-	2	1	1	2	2
Simons (1980)	C III	8	P, N	TC	-	-	1	2	0	0	0
Radtke & Straskraba (1980) OPTIMAS	S I	n (S)	P	TC	-	-	n (S)	1	0	0	0

(4) some form of model testing: verification as well as validation (see Jørgensen 1986).

These observations are in accordance with those of Straskraba and Gnauck (1985) as well as Beck (1978, 1983). If the problem and the ecosystem characteristics are not considered, it is hardly possible to select the right complexity of the model.

The conceptual diagram of the model is used as a tool to visualize the relationships of the processes, the state variable and the forcing functions. If the model is just a little complex it is impossible to test all possible parameter combinations, and it is therefore necessary to use some sort of parameter estimation methods to limit the number of parameter combinations. Finally, it must be considered absolutely necessary to test (validate) all models independent of the model development to assure the model applicability.

The selection of model complexity is a crucial problem. Jørgensen (1983, 1986) indicates that it is required to find a balance between the scope of the model and the number of observations on the one hand, and the detailed knowledge of the system and the model realism on the other. Straskraba and Gnauck (1985) claim that a higher number of state variables is not necessarily a guarantee of realistic simulations, but that the informative potential of models could be enhanced by considerations of feedback mechanisms. This is in accordance with Jørgensen (1986), where it is pointed out that a good process knowledge is needed and lack of knowledge may be overcome by studies in situ or in the laboratory.

Parameter estimations are related to the selection of complexity, as only a limited number of parameters may be determined by a calibration procedure. Furthermore, the number of observations determine the number of paramaters that can be estimated and the lower the number of parameters in the model, the less complex is needed to build the model. Some parameters can be determined by in situ or laboratory studies, but it may be concluded that the quality of the parameter estimation is one of the important factors that determine the selection of the complexity. Another important aspect of the parameter estimation is the accordance between the frequency of observations and the dynamics of the system (Jørgensen 1983, Jørgensen et al. 1986). The data that are used for calibration should always be carefully collected with a frequency corresponding to the dynamics of the calibrated system or subsystem. In Jørgensen et al. (1981) it is demonstrated how an intensive measuring period may be used to estimate some of the crucial parameters in a eutrophication model and to decide which among several process equations should be selected.

Our knowledge on process descriptions and their related parameters has been improved considerably during the last 17 years, as it can be seen in Jørgensen (1983a), in part 11 "Modelling of sub-systems" in Straskraba and Gnauck (1985) and in EPA (1985). However, as pointed out by Straskraba and Gnauck (1985), further knowledge is needed about the function of higher trophic levels, processes in the benthal region and algal sedimentation.

In the above-mentioned references several different equations for the same process may be found. This raises, of course, the question: which equation to select for a given model and a given case study? A general answer to this question cannot be given, but a more detailed description is of course needed, the more sensitive the state variable in focus (for a eutrophication model it is the phytoplankton concentration) is to the process. However, it was found by di Toro (1980) that in some cases (mainly at high nutrient concentration) it seems necessary to use the variable nutrient ratio in phytoplankton (compare also with Table 1), and by Kamp-Nielsen (1980) that in shallow lakes with low retention time of the water a more detailed description of the sediment-water exchange processes is needed. These observations are in accordance with those mentioned in Jørgensen (1976) and Jørgensen et al. (1979, 1981).

The biocoenotic structure and even the function of an ecosystem are changing during the transition from one state to another. Some very first attempts have been made to account for these changes in the models (Radtke and Straskraba 1980, Fontaine 1981, Jørgensen and Mejer 1979, 1981a, b). Experience indicates that the attempts seem promising, but far more experience is needed before what are named next generation models (Jørgensen 1981) have been fully developed.

3. TRENDS IN DEVELOPMENT OF EUTROPHICATION MODELS

The development of eutrophication models has followed several lines.

The development from CLEAN to CLEANER to MS CLEANER (see B in Table 1) presents a reductionistic approach, whereby a more and more complex model should be able to give more and more accurate descriptions and predictions, and simultaneously increase the generality of the model. As mentioned in Jørgensen (1983), improvements have evidently resulted from this development, as wide experience has been gained by this method, but it cannot be concluded that a more accurate or a more general model has been achieved.

Another line of development has aimed at the incorporation of more feedback mechanisms and description of structural changes. At present, it is hardly possible to make some unambiguous conclusions about this holistic approach, but as mentioned in Straskraba and Gnauck (1985) and Jørgensen (1986), there are some promising results.

Some of the principles of quantum mechanics have been in a way silently and slowly introduced and accepted in ecological modelling during the last 17 years.

An ecosystem is too complex to allow us to define the number of observations that are needed to set up a very detailed model - even if we still consider models with a complexity far from the complexity of nature. This is discussed in Jørgensen (1986).

It is expressed by Niels Bohr as follows: It is not possible to make one unambiguous picture (a model) of reality, as the uncertainty limits our knowledge.

The uncertainty lies in the nuclear world caused by the inevitable influence of the observer on the atomic particles and the uncertainty in ecology is caused by the enormous complexity and variability.

No map (model) of the reality is correct. There are, for instance, many maps (models) of the same land - one is used by a pilot, another one by a cardriver, a third one by a geologist, and so on. The various models reflect different viewpoints and purposes and it is not possible to decide which is the better one.

A large number of eutrophication models have been developed since 1970. They all contribute to a picture of the eutrophication process in aquatic ecosystems, but no one gives an unambiguous picture of the eutrophication even in one particular ecosystem.

This, on the other hand, does not imply that all eutrophication models contribute equally to the understanding of eutrophication. Constanza and Sklar (1985) have attempted to rate and rank models of freshwater wetland (Jørgensen 1986).

The implication of Constanza and Sklar's work is that there is an optimum size or complexity (articulation) of a model beyond which the benefits of additional articulation are outweighed by the costs of lowered accuracy.

Figures 1 and 2 summarize the results of Constanza and Sklar (for further explanation of the concepts, see Jørgensen 1986, and Constanza and Sklar 1985).

The model examined in this work is indicated in both figures. As seen, it has a high effectiveness. The descriptive accuracy is high, but the articulation (complexity) is less than the optimum. It is in accordance with the strategy applied in the development of this model: additional state variables were not added to the model, unless a good process knowledge and good observations were available.

Another parallel in the development of ecological modelling may be drawn to the differences between reducible and irreducible systems (Wolfram 1984a, b).

Physics have traditionally focussed on reducible systems, for which a simple and superior description can be given. In accordance with Wolfram this may be an exception rather than a rule for natural systems, in particular for biological systems. For irreducible systems we may know the rules, but we cannot survey the consequences of rules or make predictions, unless we

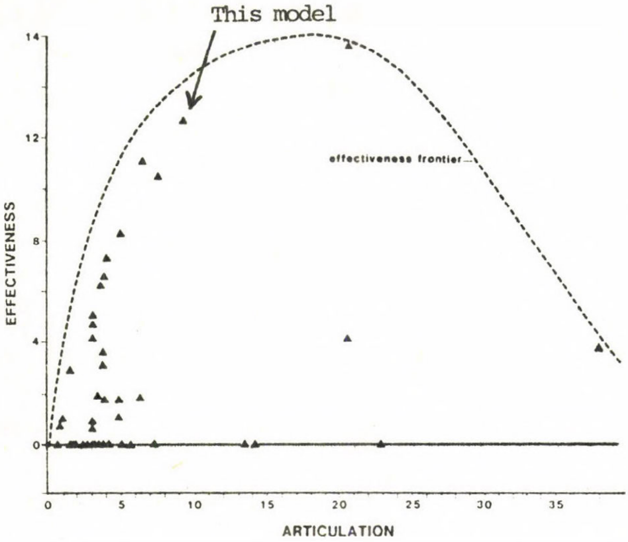


Fig. 1. Plot of articulation index vs. effectiveness index showing the current effectiveness frontier

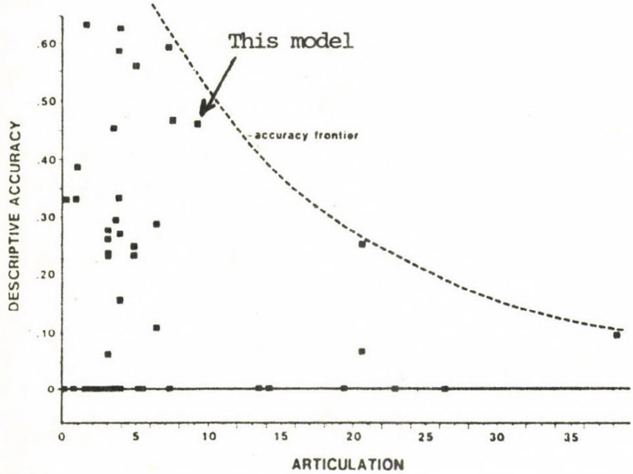


Fig. 2. Plot of articulation index vs. descriptive accuracy index for the models reviewed in this study, showing the current accuracy frontier

compute step by step how they develop. There may be simple rules for the evolution, for example, but we can only decide on the development of biological evolution by application of every step that the evolution takes. The biological evolution may be carried out by use of a computer containing all information about the living systems on earth, which, however, is never possible to obtain.

For the irreducible systems the computer becomes an experimental tool, which allows us to test consequences of rules or natural laws and compare these consequences with our observations (Wolfram 1984a, b).

The great number of ecological models including eutrophication models developed during the last 17 years represent the use of experimental mathematics in ecology. They may all have contributed to a better knowledge of the ecosystem (eutrophication models for a better understanding of the eutrophication process in aquatic ecosystems), as they have all tested some rules (laws, equations), and the results bring us a smaller or larger step forward. The method developed by Constanza and Sklar (1985) mentioned above is an attempt to evaluate the length of the step.

However, it is concluded that there are many more steps to take. Our knowledge about (1) the combination of hydrology and biology in aquatic systems; (2) higher trophic level processes; (3) feedback mechanisms; (4) benthic processes and, last but not least, (5) the structural changes including holistic approaches to the feedback mechanisms on the system level needs to be enhanced considerably before we reach the ultimate goals set up at present (Jørgensen 1983, 1986, Straskraba and Gnauck 1985).

4. THE STATE-OF-THE-ART AND CONCLUSIONS

In spite of the shortening mentioned in section 3, which should not be forgotten, the practical application of eutrophication models has shown that they can be used as a tool in environmental management including the setup of the prognosis

(Jørgensen et al. 1986), provided that the recommendations given above are followed.

The models developed have furthermore a certain but not complete generality but a good knowledge of the characteristics of the ecosystem is needed to make the required changes from case to case.

Research has been initiated to eliminate the shortcomings and in particular to be able to predict structural changes and shifts in species composition. This should be expected to be the next great step forward in the development of reliable eutrophication models in environmental management and research.

We do know how to produce an acceptable eutrophication, but we need further progress before we have an excellent model in hand. In any case, we are optimistic.

REFERENCES

Beck, M.B. (ed.) (1978): Mathematical modelling of water quality. Summary Report of a IIASA Workshop, September 13-16, 1977. Collaborative Proceedings CP-78-10, Laxenburg, Austria: International Institute for Applied Systems Analysis.

Beck, M.B. (1983): A Procedure for Modelling in Mathematical Modelling of Water Quality: Streams, Lakes and Reservoirs, edited by G.T. Orlob, IIASA. John Wiley, New York, pp. 235-255.

Chahuneau, F., Desclers, S., Meyer, J.A. (1980): Les modèles de simulation en ecologie lacustre. Présentation des différentes approches et analyse des modèles existant. Acta Oecologica I, 27-50.

Constanza, R., Sklar, F.H. (1985): Articulation, accuracy and effectiveness of mathematical models: a review of freshwater wetland applications, Ecol. Modelling 27, 45-69.

EPA (1985): Rates, Constants, and Kinetic Formulations in Surface Water Quality Modeling (Second Edition).

Fontaine, T.D. (1981): A self-designing model for testing hypotheses of ecosystem development. In: D. Dubois (ed.): Progress in Ecological Engineering and Management by Mathematical Modelling.

Jørgensen, S.E., Mejer, H. (1979): A holistic approach to ecological modelling. Ecol. Modelling, 7, 169-189.

Jørgensen, S.E. (1981): A holistic approach to ecological modelling by application of thermodynamics. In: W. Mitsch, R.K. Bosserman, J.A. Dillon, Jr. (eds): Systems and Energy 1982, Ann Arbor Science.

Jørgensen, S.E., Mejer, H.F. (1981a): Application of energy in ecological models. In: D. Dubois (ed.): Progress in Ecological Modelling, Liège, pp. 311-347.

Jørgensen, S.E., Mejer, H.F. (1981b): Energy as key function in ecological models. In: W. Mitsch, R.W. Bosserman, J.M. Klopatek (eds): Energy and Ecological Modelling. Developments in Environmental Modelling 1. Elsevier, Amsterdam-Oxford-New York, pp. 587-590.

Jørgensen, S.E. (1986): Fundamentals of Ecological Modelling, 9. Elsevier, Amsterdam.

Jørgensen, S.E. (1983): Ecological Modelling of Lakes. In: G.T. Orlob (ed.): International Series on Applied Systems Analysis, 12. J. Wiley, New York.

Jørgensen, S.E. (1983a): Modelling the Ecological Processes. In: G.T. Orlob (ed.): International Series on Applied Systems Analysis, 12. J. Wiley, New York.

Jørgensen, S.E., Jørgensen, L.A., Kamp-Nielsen, L., Mejer, H.F. (1981): Parameter Estimation in Eutrophication Modelling. Ecological Modelling, 13.

Jørgensen, S.E. et al. (1986): Validation of a prognosis based upon a Eutrophication Model. Ecological Modelling, 32.

Jørgensen, S.E., Kamp-Nielsen, L., Jørgensen, L.A. (1986): Examination of the Generality of Eutrophication Models. Ecological Modelling, 2.

Jørgensen, S.E. (1986): Structural Dynamic Model. Ecological Modelling, 31.

Kamp-Nielsen, L. (1980): The influence of sediments on changed phosphorus loading to hypertrophic L. Glumsø. In: J. Barica, L.R. Mur (eds): Developments in Hydrobiology, Vol. 2. Junk, Publ., The Hague, The Netherlands.

Radtke, E., Straskraba, M. (1980): Self-Optimization in a Phytoplankton Model. Ecol. Modelling, 9, 247-268.

Straskraba, M., Gnauck, A.H. (1985): Freshwater Ecosystems. Modelling and Simulation. Developments in Environmental Modelling 8. Elsevier, Amsterdam.

Toro, D.M. di (1980): Applicability of cellular equilibrium and monod theory to phytoplankton growth kinetics. Ecol. Modelling 8, 201-218.

Wolfram, S. (1984a): Computer software in science and mathematics. Scientific American 251, 140-151.

Wolfram, S. (1984b): Cellular automata as models of complexity. Nature 311, 419-424.

LIST OF PARTICIPANTS

ABAFFY-BOTHÁR, Anna
Hungarian Danube Research Station, Jávorka Sándor u. 14, 2131 Göd, Hungary

ALLAN, R.
Canada Centre for Inland Waters, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

ANDERSEN, S.
U.S. Environment Protection Agency, 401 M St. SW, Washington DC 20460, U.S.A.

ANTAL, G.
Central-Transdanubian District Water Authority, Balatoni u. 6, 8000 Székesfehérvár, Hungary

AYIBOTELE, N.B.
Water Resources Research Institute, Ghana

BARICA, J.M.
National Water Research Institute, Canada Centre for Inland Waters, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

BAUER, C.E.
World Federation of Engineering, Balcare 226-3 P., 1064 Buenos Aires, Argentina

BÍRÓ, P.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

BLOM, G.
Agricultural University, Dept. of Water P., De Dreijen 12, 6703 BC Wageningen, The Netherlands

BONIFER Sz., Márta
Ministry for Environment and Water Management, Fő u. 48-50, 1011 Budapest, Hungary

BORBÉLY, Gy.
Szeged Biological Center of the Hungarian Academy of Sciences, Odesszai krt. 62, P.O. Box 521, 6701 Szeged, Hungary

BROWN, A.
Nature Conservancy Council, 12 Hope Terrace, Edinburgh EH9 2AS United Kingdom

BUGENYI, F.
Uganda Freshwater Fisheries Research Organization, P.O. Box 343, Jinja, Uganda

CARILLO, P.
University of Granada, Faculty of Sciences, Avda-Fuentenueva, 18071 Granada, Spain

CASPER, J.
Academy of Sciences Zimet/Limnology, Beutenberger Str. 11, 6900 Jena, GDR

CHANG, P.S.
Freshwater Institute, Department of Fisheries, 501 University Crescent, Winnipeg R3T 3S5, Canada

CHARLTON, M.
National Water Research Institute, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

CLAASSEN, T.
Provincie Friesland, P.O. Box 20120, Leenwarden 8900 HM, The Netherlands

CSÁNYI, B.
Research Centre for Water Resources Development (VITUKI), Kvassay Jenő út 1, 1095 Budapest, Hungary

CSERNY, T.
Hungarian Geological Institute, P.O. Box 106, 1442 Budapest, Hungary

DATTA, N.C.
Department of Zoology, University of Calcutta, 35 B.C. Road, 700 019 Calcutta, India

DEMARTA, Antonella
Istituto Cantonale Batteriosierologico, Via Ospedale 6, CH-6900 Lugano, Switzerland

DIAMANT, R.
U.S. National Park Service, 99 Warren, Brookline MA 02146, U.S.A.

DIÓSI, A.
Ministry for Environment and Water Management, Fő u. 48-50, 1011 Budapest, Hungary

DOBOLY, E.
Research Centre for Water Resources Development (VITUKI), Kvassay Jenő út 1, 1095 Budapest, Hungary

DOBSON, T.
Michigan State University, East Lansing, MI 1244 Cedarhill, 48823, U.S.A.

DOKULIL, M.
Institute of Limnology OAW, Gaisberg 116, 5310 Mondsee, Austria

DOWDESWELL, E.
Great Lakes Environment Office, Conservation and Protection of Ontario Region Environment, 25 St. Clair Avenue East, Toronto, Ontario M4T 1M2, Canada

DU HENGCHAN
Sichuan Provincial Construction Committee, Chengdu, 31, Xiyujie, China

EGGEREIDE, A.
Directorate for Nature Management, Tungasletta 2, 7015 Trondheim, Norway

EKHOLM, P.
National Board of Waters and Environment, P.O. Box 250, SF 00101 Helsinki, Finland

FINCH, Carol
Great Lakes National Program Office of the U.S. EPA, 230 South Dearborn Street, Chicago, Illinois 60604, U.S.A.

FORSBERG, C.
Institute of Limnology, P.O. Box 557, 75122 Uppsala, Sweden

FRISK, T.
National Board of Waters and Environment, P.O. Box 250, SF 00101 Helsinki, Finland

GALAZY, G.
Ecological Museum of the USSR Academy of Sciences, 66016 Irkutsk, Lystvyanka, USSR

GERULAITIS, A.
Institute of Zoology and Parasitology, Lithuanian SSR, Vilnius, USSR

GODA, T.
Setsuan University, Ikeda-nakamachi 17-8, Heyagawa, Osaka 572, Japan

GOLDMAN, C.
Division of Environmental Studies, University of California, Davis, California 95616, U.S.A.

GOLTERMAN, H.L.
Station Biologique de la Tour du Valat, F-13200 Arles, France

GOLUBEV, G.N.
United Nations Environment Programme, P.O. Box 30552, Nairobi, Kenya

GÓMEZ CARRO, R.
Moron Ciego de Avila, Salome Machado No. 4, Cuba

GOPAL, B.
School of Environmental Sciences, New Delhi 110067, India

GROSHEVA, E.I.
Institute of Ecological Toxicology, P.O. Box 48, Baikalsk, USSR

GU, D.
Nanjing Institute of Geography and Limnology, Academia Sinica, 73 East Beijing Rd., Nenjing, China

GUNKEL, G.
Technical University Berlin, Str. des 17. Juni 135, 1000 Berlin (West)

HAMAR, J.
Regional Hydrobiological Laboratory, P.O. Box 32, 5000 Szolnok, Hungary

HERODEK, S.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

HESTHAGEN T.
Directorate for Nature Management, Fish Research Division, Tungasletta 2, 7004 Trondheim, Norway

HILLBRICHT-ILKOWSKA, A.
Institute of Ecology, Polish Academy of Sciences, Dziekanów Lesny, 05-092 Łomianki, Poland

ILMAVIRTA, V.
Maj and Tor Nessling Foundation, P. Hesperiankatu 3 A 4, SF-00260 Helsinki, Finland

IMAI, K.
ILEC Secretariat, 4-1-1 Kyomachi, Otsu, Shiga 520, Japan

INABA, M.
Shiga Prefecture 4-1-1 Kyomachi, Otsu, Shiga 520, Japan

INFANTE, A.
Apartado 47106 Los Chaguaramos, Caracas 1041, Venezuela

ISTVÁNOVICS, Vera
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

JANA, B.B.
Fisheries and Limnology Research Unit, Department of Zoology, University of Kalyani, Kalyani-741 235, West Bengal, India

JANURIK, E.
Fisheries Research Institute, P.O. Box 47, Szarvas, Hungary

JESZTÁK, Á.
South-Transdanubian District Water Authority, Köztársaság tér 7, 7623 Pécs, Hungary

JØRGENSEN, S.E.
Royal Danish School of Pharmacy, Department of Pharmaceutical Chemistry, 2 Universitetsparken, DK-2100 Copenhagen, Denmark

KEREKES, J.
Dalhouse University, Canadian Wildlife Service, Halifax N.S. B3H 4JI, Canada

KETTUNEN, V.E.J.
Laboratory of Hydrobiology and Water Resources Engineering, Helsinki University of Technology, Rakentajanaukio 4, SF-02150 Espoo, Finland

KHALAF AZWAR
Biological Research Centre, Scientific Research Council, P.O. Box 2371, Jadriyah, Baghdad, Iraq

KIRA TATUO
Lake Biwa Research Institute, 1-10 Uchide-hama, Otsu, Shiga 520 Japan

KISMUL, V.
State Pollution Control Authority, P.O. Box 8100, Oslo, Norway

KLESSIG, L.
College of Natural Resources, University of Wisconsin, Stevens Point WI 54481, U.S.A.

KLOKK, T.
Directorate for Nature Management, Tungasletta 2, 7004 Trondheim, Norway

KONCSOS, L.
Research Centre for Water Resources Development (VITUKI), Kvassay Jenő út 1, 1095 Budapest, Hungary

KORNAKOVA, E.F.
Ecological Museum, Irkutsk District, 666016 Listvyanka, USSR

KOSCHEL, R.
Academy of Sciences of the GDR, Central Institute of Microbiology and Experimental Therapy, Department of Limnology, Beutenberg Str. 11. 6900 Jena, GDR

KOTANI, H.
ILEC Secretariat, 4-1-1 Kyomachi, Otsu, Shiga 520, Japan

KOVÁCS, Á.
South-Transdanubian District Water Authority, Köztársaság tér 7, 7623 Pécs, Hungary

KŐVÁRI-GULYÁS, E.
Engineering Advisory Board, Reviczky u. 4, 1088 Budapest, Hungary

KURATA, A.
Lake Biwa Research Institute, 1-10 Uchide-hama, 520 Otsu, Shiga, Japan

KUTAS, T.
Computer and Automation Institute of the Hungarian Academy of Sciences, Kende u. 13-17, 1111 Budapest, Hungary

LAKATOS, Gy.
Institute of Ecology, Kossuth Lajos University, 4010 Debrecen, Hungary

LIEPA, R.A.
Institute of Biology, Latvian SSR Academy of Sciences, 3 Miera str., 229021 Salaspils, Latvian SSR, USSR

LINNIK, P.M.
Institute of Hydrobiology, Ukrainian SSR Academy of Sciences, Geroyev Stalingrada Ave. 12, 252210 Kiev, USSR

LIU HONGLIANG
Chinese Research Academy of Environmental Sciences, Beiyuan, Beijing, China

LIU LIU
Sichuan Learned Society of Environmental Science, 31 Xiyu Street, Chengdu, Sichuan, China

LOFFLER, H.
University of Vienna, Althan Str. 14, Vienna, Austria

LOGA, M.
Warsaw Technical University, Institute of Environmental Engineering, Nowowiejska 20, 00-650 Warsaw, Poland

MAGADZA, C.H.D.
University Lake Kariba Research Station, P.O. Box 48, Kariba, Zimbabwe, Africa

MARIAZZI, A.
Instituto de Limnologia, Facultad de Ciencias Naturales y Museo, Universidad Nacional de La Plata, Paseo del Bosque, 1900 La Plata, Argentina

MASINI, I.
Brasilia Water and Sewage Corporation, CAESD, Bloco A No. 67/97, Cuadra 04, Brasilia, Brazil

MASTRANTUONO, L.
Dipartimento di Biologia Animale e dell'Uomo, Viale dell'Università 32, 00185 Rome, Italy

MÁTHÉ, F.
Keszthely Agricultural University, P.O. Box 66, 8361 Keszthely, Hungary

MATSUI, S.
Faculty of Technology, Kanazawa University, 2-40-20 Kodatsuno, Kanazawa 920, Japan

MAVUTI, K.M.
University of Nairobi, Department of Zoology, Nairobi, Kenya

MISLEY, K.
Ministry for Environment and Water Management, Fő u. 44-50, 1011 Budapest, Hungary

MOISEENKO, T.
Kola Branch of the USSR Academy of Sciences, Apatity, USSR

MOLOT, L.A.
Ramsay Wright Zoological Laboratories, University of Toronto, 25 Harbord Str., M5S 1A1 Toronto, Ontario, Canada

MOOG, O.
University of Agriculture, Department of Hydrobiology, Fisheries and Aquaculture, Vienna, Austria

MUDRYK, Z.
Department of Ecology, Institute of Biology, Pedagogical University, Arckiszewskiego 22b, 76-200 Słupsk, Poland

MUNAWAR, M.
Canada Centre for Inland Waters, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

MURAOKA, K.
The National Institute for Environmental Studies, Onogawa 16-2, Tsukuba, Japan

MURATOWA, S.
Agricultural University, Os. Kraju Rad 13768, 61-675
Poznań, Poland

NAGY-BODOR, E.
Hungarian Geological Institute, P.O. Box 106,
1442 Budapest, Hungary

NASH, S.
International Waterfowl Research Bureau, Gloucester GL2
7DX, Slimbridge, England

NÉMETH, J.
Research Centre for Water Resources Development (VITUKI),
Kvassay Jenő út 1, 1095 Budapest, Hungary

NYBOM, C.
National Board of Waters and Environment, P.O. Box 250,
SF 00101 Helsinki, Finland

OCHUMBA, P.B.O.
Kenya Marine and Fisheries Research Institute, Kisumu
Laboratory, P.O. Box 1881, Kisumu, Kenya

OGAWA, A.
ILEC Secretariat, 4-1-1 Kyomachi, Otsu, Shiga 520, Japan

OLÁH, J.
Fisheries Research Institute, P.O. Box 47, 5541 Szarvas,
Hungary

OLRIK, Kirsten
Environmental Biology Laboratory, Bogardsvej,
3050 Humlebaek, Denmark

O. TÓTH, E.
Fisheries Research Institute, P.O. Box 47, 5541 Szarvas,
Hungary

PADISÁK, Judit
Department of Botany, Hungarian Natural History Museum,
P.O. Box 222, Könyves Kálmán krt. 40, 1476 Budapest,
Hungary

PAL, S.G.
Calcutta University Teachers' Qtrs, Flat No. C-6, PI/7
C.I.T. Scheme VII. M., Calcutta 700054, India

PARPAROV, A.S.
Sevan Hydrobiological Station, Kirov 186, 378610 Sevan,
Armenian SSR, USSR

PAUTOVA, V.N.
Institute of Ecology of the Volga River Basin of the USSR
Academy of Sciences, 445003 Togliatti, USSR

PEKAR, F.
Fisheries Research Institute, P.O. Box 47, 5541 Szarvas, Hungary

POMOGYI, Piroska
District Water Authority, Csík F. sétány 1, 8360 Keszthely, Hungary

RAO, S.
Rivers Research Branch, National Water Research Institute, Canada Centre for Inland Waters, Environment Canada, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

RASPOPOV, I.M.
Institute of Limnology, Academy of Sciences of the USSR, ul. Sevastyanova 9, Leningrad, USSR

RYCHKOV, I.L.
P.O. Box 24, Irkutsk 3, USSR

SAAD, M.A.H.
Oceanography Department, Faculty of Science, Alexandria University, Moharem Bey, Alexandria, Egypt

SAKIMURA, H.
United Nations Environment Programme, P.O. Box 30552, Nairobi, Kenya

SALABARRIA FERNANDEZ, D.M.
National Commission for Environmental Protection, Cuban Academy of Sciences, Ave. 17, No. 5008e/ 50 y 52, Playa, Habana, Cuba

SALÁNKI, J.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

SANCHEZ-CASTILLO, P.
Departamento de Biologia Vegetal, Departamento de Biologia Animal, Ecologia y Genética, Facultad de Ciencias. Universidad de Granada, 18001 Granada, Spain

SARTORIS, J.J.
U.S. Bureau of Reclamation, P.O. Box 25007-D-3742, Denver Co., USA

SCHELLENBERGER, G.
Institute of Geography and Geoecology, Academy of Sciences of the GDR, Muggelseedam 32, 1162 Berlin, GDR

SHAFIK, H.M.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

SHIMIZU, R.
Great Lakes Environment Office, Conservation and Protection Ontario Region, 6th Floor, 25th Clair Ave East, M4T 1M2 Toronto, Ontario, Canada

SIRVIÖ, H.E.
Helsinki University of Technology, Rakentajanaukio 4, SF-02150 Espoo, Finland

SISÁK, I.
Keszthely Agricultural University, Deák F. u. 16, 8360 Keszthely, Hungary

SOMLYÓDY, L.
Research Centre for Water Resources Development (VITUKI), P.O. Box 27, 1453 Budapest, Hungary

STRASKRABA, M.
Biomathematical Laboratory, Southern Bohemian Biological Research Centre, Czechoslovak Academy of Sciences, Branisovská 31, 370 05 České Budejovice, Czechoslovakia

SYMOENS, J.J.
Laboratorium voor Algemene Plantkunde en Natuurbeheer, Vrije Universiteit Brussel, Pleinlaan 2, 1050 Brussels, Belgium

SZELIGIEWICZ, W.
Institute of Paleobiology, Zwirki i Wigury 93, 02-089 Warsaw, Poland

SZENES, E.
Ministry for Environment and Water Management, Fő u. 48-50, 1011 Budapest, Hungary

SZILÁGYI, F.
Research Centre for Water Resources Development (VITUKI), P.O. Box 27, 1453 Budapest, Hungary

SZUMIEC, Maria Anna
Experimental Fish Culture, Stat. Golysz 43-422 Chybie, Poland

TATEKAWA, M.
Bukkyo University, 43 Tojihigashi-monzencho, Kyoto 601, Japan

TEREK, J.
Branch for Ecology of Agricultural Landscape, CBES SAS, Kukorelliho 10, 04000 Košice, Czechoslovakia

TONOLLA, M.
Istituto Cantonale Batteriosierologico, Via Ospedale 6, CH-6900 Lugano, Switzerland

TROJANOWSKA, Czeslawa
Department of Chemistry, Institute of Biology, Pedagogical University, Arciszewskiego 22, 76-200 Słupsk, Poland

TROJANOWSKI, J.
Department of Chemistry, Institute of Biology, Pedagogical University, Arciszewskiego 22, 76-200 Słupsk, Poland

TURCSÁNYI, G.
Agricultural University, Páter Károly u. 1, 2103 Gödöllő, Hungary

VAN DAM, H.
Research Institute for Nature Management, P.O. Box 46, 3956 ZR Leersun, The Netherlands

VARANKA, I.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

VARIS, O.
Helsinki University of Technology, Rakentajanaukio 4, SF-02150 Espoo, Finland

VASILEV, O.F.
Institute of Water and Environmental Problems, Morskoy Prospect 2, 630090 Novosibirsk 99, USSR

V.-BALOGH, Katalin
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

VOLLENWEIDER, R.A.
National Water Research Institute, Canada Centre for Inland Waters, P.O. Box 5050, L7R 4A6 Burlington/Ontario, Canada

VÖRÖS, L.
Balaton Limnological Research Institute of the Hungarian Academy of Sciences, P.O. Box 35, 8237 Tihany, Hungary

VRHOVSEK, D.
Institute of Biology, P.O. Box 141/3, 61000 Ljubljana, Yugoslavia

WALLSTEN, Maud
Institute of Limnology, P.O. Box 557, S-751 22 Uppsala, Sweden

WATANABE, M.M.
The National Institute for Environmental Studies, 16-2 Onogawa, Tsukuba, Ibaraki 305, Japan

WEDEPOHL, R.
Department of Natural Resources, Madison, WI 53706, U.S.A.

WEYENETH, R.R.
University of Hawai, Department of American Studies, Honolulu, 96822 Hawai, U.S.A.

WIŚNIEWSKI, R.
Institute of Biology, University of N. Copernicus, ul. Gagarina 9, 87-100 Toruń, Poland

XIONG SIYU
Standing Committee of Kunming Municipal People's Congress, City and Township Commission, East Dongfeng Rd., Kunming/Yunnan Province, China

XIONG YUE
Kunming Municipal Environmental Protection Bureau, 51-1 Xin-wen Road, Kunming, China

YAMAZAKI, K.
ILEC Secretariat, 4-1-1 Kyomachi, Otsu, Shiga 520, Japan

ZHANG JIQIANG
Chinese Research Academy of Environmental Sciences, Beiyuan, Beijing, China

ZOLTAN, S.
Affiliated Candlewood Watersheed, Brookfield, 06804 CT Candlewood Orchards, U.S.A.

ŻYTKOWICZ, R.
Department of Hydrobiology, University of N. Copernicus, ul. Gagarina 9, 87-100 Toruń, Poland

SUBJECT INDEX

The page numbers refer to the first page of the paper in which the index term appears.

HEAVY METALS IN WATER ORGANISMS

Proceedings of a Symposium held at the Balaton Limnological Research Institute of the Hungarian Academy of Sciences in Tihany, Hungary

Edited by
J. SALÁNKI

(Symposia Biologica Hungarica 29)

Papers in English. 1985. XVI + 440 pages. 123 figures, 86 tables. 17 × 25 cm. Hardbound
ISBN 963 05 4195 5

Heavy metals have come into the focus of environmental biological research all over the world. Beyond the monitoring of pollution, special attention is being paid to the various organisms which make early detection possible.

The papers in this volume were presented at a symposium, where scientists from 12 countries provided information on the latest results in this field. The most important topics discussed were accumulation of heavy metals, ecological monitoring of heavy metal pollution, organisms as indicators and the effect of heavy metal pollution on vital functions. The Hungarian participants gave an account of the state-of-the-art of research into the heavy metal pollution of Lake Balaton for the first time.

The seriousness of the problem of heavy metal pollution is reflected by the fact that it has assumed prominence in one of the main projects of IUBS.

DISTRIBUTOR

KULTURA

Hungarian Foreign Trading Company
P.O.Box 149
H-1389 Budapest, Hungary